POTENTIAL BIOLOGICAL CONSEQUENCES OF SUBMARINE MINE-TAILINGS DISPOSAL: A LITERATURE SYNTHESIS

Prepared by
Edward R. Kline

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Prepared for
U.S. Bureau of Mines
Alaska Field Operations Center
P.O. Box 20550
Juneau, Alaska
99802-0550
907-364-2111
Fax: 907-364-3622

Prepared by
Edward Kline
Juneau Center, School of Fisheries and Ocean Sciences
University of Alaska Fairbanks
and University of Alaska Southeast
Juneau, Alaska

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POTENTIAL BIOLOGICAL CONSEQUENCES OF SUBMARINE MINE TAILINGS DISPOSAL: A LITERATURE REVIEW

By Edward Kline[1]

ABSTRACT

A review and synthesis of literature pertaining to biological consequences of submarine mine-tailings disposal (STD) was conducted. STD can result in massive sea floor sediment deposition. STD may also increase suspended sediment, trace metals, and residual milling reagents in receiving waters. These perturbations, which are highly site dependent, invariably smother benthic organisms, and could potentially affect or alter fish, plankton, and benthos through acute and chronic toxicity, bioaccumulation, behavioral changes, smothering, derived secondary effects, and habitat alteration.

Much of the available information concerning STD is unpublished and limited to a restricted number of sites. From this information and related research, it appears that benthic smothering is the only major consequence of a properly designed STD system. There have been cases of metal bioaccumulation due to inadequate preliminary evaluation. Also, shallow water habitat alteration has resulted from STD. Reductions in biological production are likely due to benthic smothering. The rate of ecological recovery after termination of an STD operation varies and is difficult to assess.

The consequences of STD are site specific. Additional research is needed concerning possible consequences of milling reagents and the broad implications of benthic smothering. Also, methodologies are needed to allow better prediction of benthic recolonization and ecological implications of metal bioaccumulation.

[1] Juneau Center, School of Fisheries and Ocean Sciences, University of Alaska Fairbanks, and University of Alaska Southeast, Juneau, AK

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1.0 INTRODUCTION

One of the major challenges for a metal mining operation is disposal of the spent tailings in a cost effective and environmentally responsible manner. Foremost among environmental concerns is the introduction of toxic metals to aquatic ecosystems. Oxidation and subsequent metal leaching are minimized through submersion of tailings in water. Aside from the often impractical alternatives of dry disposal or mine back filling, this leaves two options in a coastal mining operation, on-land submersion or submarine tailings disposal (STD).

On-land tailings disposal often involves costly construction of an impoundment to flood the tailings, raising concerns over the obvious or potential impacts on aesthetics, recreation, wildlife, and surface and ground water quality. Also, any dissolved metals or undegraded milling reagents in the spill-over water, if not treated, will be released to the sea regardless (USBLM 1991, USDA 1992). Finally, on-land impoundments create the long-term responsibility and expense of maintenance to avoid catastrophic flooding and contaminant release.

In comparison, STD should result in temporary impacts, and is a permanent means to dispose of tailings. Given the proper circumstances, it is often preferred over on-land disposal from an environmental and economic view (Echo Bay Mgmt Co. 1988, Hesse and Reim 1993). The economic advantages to the mine operator, particularly during construction and reclamation, are self evident with regard to dam construction and maintenance.

Numerous biotic and abiotic factors must be considered when evaluating potential STD sites (Poling et al. 1993, Ellis et al. in press). These factors include ore composition, bathymetry, current velocities, and resident biota (Hesse and Reim 1993). Ideally, the ore should be of low reactivity and the tailings released into a deep basin with low current velocities, high natural sedimentation, and low biological productivity.

In properly engineered STD systems, particularly in British Columbia, reported environmental impacts have been minimal (Poling et al. 1993), though there have been instances involving resuspension of deposited tailings (Goyette and Nelson 1977). Another example that warrants caution is a former STD operation in Greenland at which there was an appreciable release of metals from reactive tailings and near shore waste rock, and subsequent metal accumulation in biota (Asmund et al. 1991).

Current regulations prohibit discharge of flotation mill process wastewaters to the waters of the United States (40 CFR 440.102) (McDonald and Martin 1992, Hesse and Reim 1993). Because mill wastewater is unavoidably associated with some of the tailings, and froth floatation is the only feasible method of metal extraction for U.S. mines that could potentially use STD (USBLM 1991, USDA 1988a, USDA 1988b, USDA 1992), marine discharge is currently not an option in this country. Scientific justification for the U.S. prohibition has not been forthcoming.

The only state likely to benefit from STD is Alaska, due to abundant coastal mineral reserves combined with its unique near shore topography. Metals accounted for 81% of Alaskan mineral production in 1992 (Swainbank et al. 1993) and some feel the state is poised for a mining boom (Walker 1991). The U.S. Bureau of Mines has identified 20 sites, ranging from the Alaska Peninsula to Ketchikan, where preliminary screening has indicated a coincidence of potential mineral reserves with potentially suitable STD conditions (Coldwell and Gensler 1993).

The literature concerning STD lacks detailed laboratory and field studies addressing specific biological aspects of STD. It is also largely in-house, the so called "gray literature", rather than in peer-reviewed scientific journals. The U.S. Bureau of Mines (Baer et al. 1992) has compiled a bibliography of published and unpublished literature relating to STD. A review of the reports that are available revealed limited research addressing the consequences of STD to marine biota. Rather, potential consequences must largely be inferred from related research concerning dredging, deep-sea mining, and disposal of other industrial effluent.

To some degree, STD results in increases of suspended sediment, trace metals, residual milling reagents, and sediment deposition in receiving waters. These components could potentially affect or alter benthic associated invertebrates and fish, pelagic fauna including zooplankton and fish, and the phytoplankton and microbiota on which these animals rely. STD creates the potential for acute and chronic toxicity, bioaccumulation, behavioral changes, smothering, secondary or indirect effects, and habitat alteration.

1.1 Intent of Review

The intent for this review is to synthesize the available biological literature, directly or indirectly related to STD. The focus will be on potential large scale biological changes considered important from an ecological, recreational, or commercial stand point, implying fish, invertebrates, and the organisms on which they rely. Potential effects on marine mammals will not be addressed, though it is a subject worthy of consideration. General conclusions regarding the impacts of STD on biota will be drawn, and areas requiring further research will be defined.

1.2 Acknowledgements

I thank Roger Baer for the opportunity to prepare this report, and Michael Stekoll, of the University of Alaska; Pamela Drake, of the U.S. Bureau of Mines; Derek Ellis, of the University of Victoria; and Scott Johnson, of NOAA's National Marine Fisheries Service, for their thorough and constructive critiques.

2.0 DESCRIPTION OF SUBMARINE TAILINGS DISPOSAL

The following is a general description of a model STD system. The conclusions of this report will be in reference to operations fitting this general description, though details may vary considerably between operations.

2.1 Tailings Slurry Composition and Discharge

The material that is discharged during STD consists of a slurry of tailings and mill wastewater. During the milling process, target metals are extracted through a series of physical and chemical procedures resulting in solid waste typically reduced to clay ($<4\ \mu m$ diameter) through sand (ca 0.5 mm) sized particles. The associated wastewater may contain trace metals which have been mobilized in the milling process. Also in the wastewater are residual reagents added to separate and concentrate the valuable metals, and possibly reagents added to increase the flocculation of the tailings upon discharge.

Poling et al. (1993) detail the physical features of a model STD system (Figure 2-1). Upon leaving the mill, the tailings slurry is piped to a sea water mixing and deaeration chamber which serves to increase the density of the fluid and minimize sediment suspension. It is discharged generally at depths between 50 and 150 m. In the near-field zone, defined as a 100 m radius from the point of discharge, there may be a plume of suspended sediment and wastewater. The majority of the tailings travel down-slope in the form of a turbidity current (Middleton and Hampton 1976). Over riding this cohesive flow is a suspension of fine particles varying in depth and concentration.

The depositional depths of STD waters typically range between 100 and 400 m. The behavior of the turbidity current once it reaches the basin bottom is likely to take the form of a meandering channel, a spreading apron, or alternating between the two (Hay 1982, Hay et al. 1983) resulting in deposits that can be 100 m thick. Over the course of an STD operation, generally at least 10 years, the entire width of a fjord or inlet bottom may be covered by tailings with approximately 10% of the deposit undergoing active deposition at any one time. Barring resuspension, the remainder of the deposit, to some degree, will be undergoing natural sediment burial.

2.2 Components of Potential Biological Consequence

Chemical and physical components of STD may contribute to biotic effects or changes. The chemical components are trace metals and residual milling reagents. Physical components include increased suspended sediment and sediment deposition. The possibility for increased suspended sediment exists in the entire water column, particularly in the near-field plume and at depth. Increased trace metal concentrations are also possible throughout the receiving waters, particularly near the bottom. Residual milling reagents will most likely only be appreciable in the near-field plume due to dilution and/or degradation beyond this area (Kessler/E.V.S. 1992). Sediment deposition is, of course, a concern on the bottom of the basin through instantaneous deposition from the turbidity current and more gradual deposition from slowly sinking suspended tailings on the periphery of the primary deposit.

It should be noted that the effects and bioaccumulation potential of substances in combination cannot be predicted based on their individually determined values. The presence of one compound (i.e. metals, milling reagents, suspended sediment) may influence the biological activity of another in an additive, synergistic, or antagonistic manner.

2.2.1 Suspended Sediment

Suspended sediment concentrations vary greatly in coastal areas depending largely on depth, wave energy, tidal currents, and proximity to rivers (Drake 1976, Komar 1976, Swift 1976). The settling velocity of sediment depends on grain size, shape, density, flocculation, and water turbulence (Dyer 1976), all of which will vary between STD operations. The depth and concentration at which suspended sediment occurs primarily determines the potential biological consequences. These include effects on primary production due to light attenuation and direct physical or behavioral effects on animals.

Some terminology in reference to suspended sediment should be clarified. While suspended sediment concentrations are often referred to as turbidity, this can be misleading. The latter is

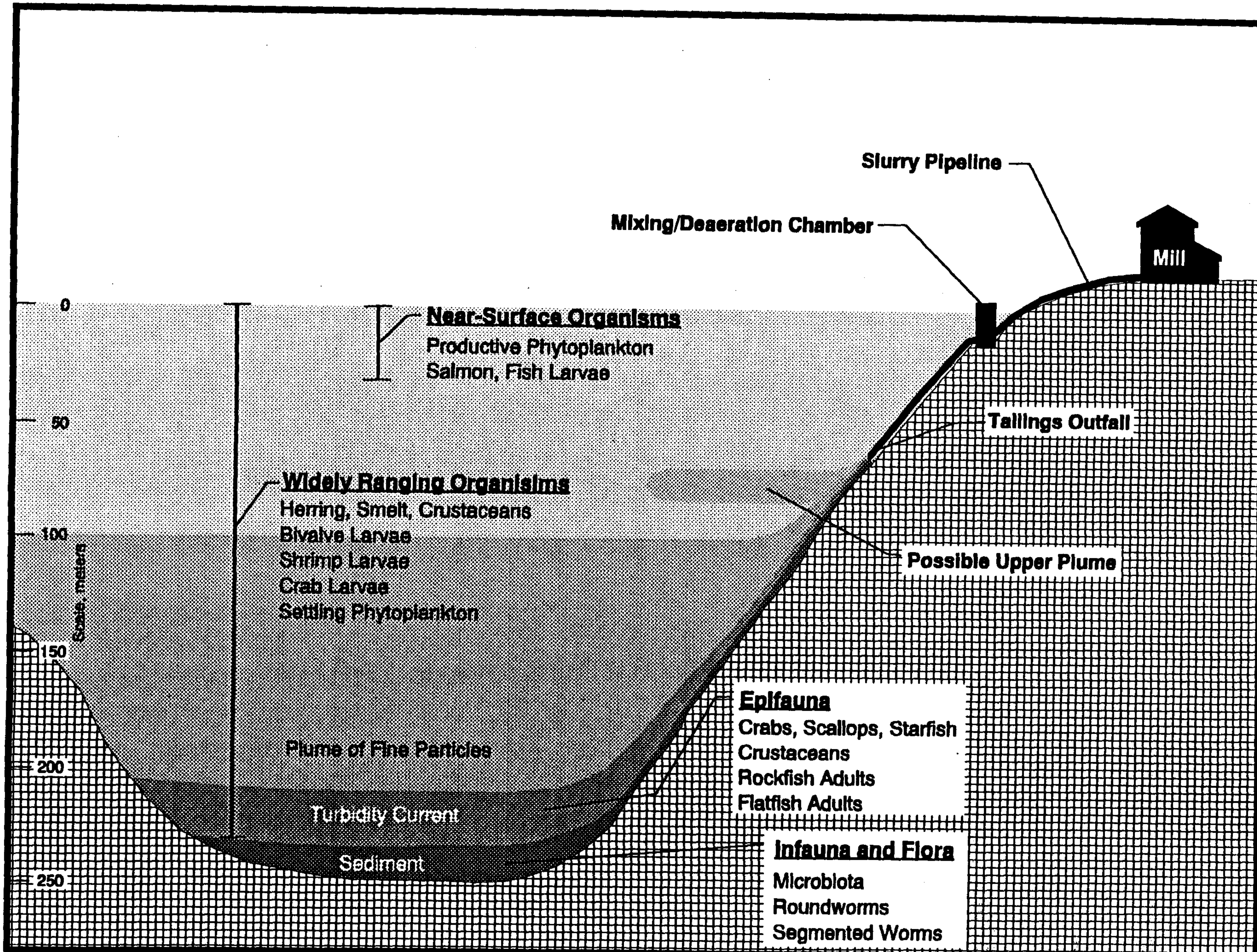

Figure 2.1 - Biota depth distributions relative to generalized submarine tailings disposal components.

a measure of light transmissivity and is affected by characteristics of the particles which make a reliable correlation to suspended sediment concentrations unlikely (Noggle 1978, Peddicord 1980). Also, while weight per unit volume is often used interchangeably with proportion of parts (e.g., parts per million) when referring to dissolved solutes, this is occasionally interchanged incorrectly in the literature when referring to suspended sediment. In this review, suspended sediment will be referred to as such and will be presented in weight per unit volume (i.e., mg/L or g/L).

2.2.2 Metals

The mobilization of metals in the milling process combined with the increased surface area exposed by reduction of rock to tailings creates the potential for dissolved metal levels exceeding natural background concentrations. The mobilization of metals during STD is most likely to occur when the tailings are in contact with oxygenated water. The degree of metal leaching is highly related to the mineralogical composition of the tailings. Tailings high in sulfides are less soluble than other forms, such as oxides, hydroxides, and carbonates (Poling et al. 1993). Deposited sediment contains the highest concentrations of metals since metals are eventually transported to the bottom in the tailings themselves, through the death of organisms which ingested or accumulated them, or through sorption to other sinking particles (Forstner and Wittmann 1979). Once buried to sufficient depth, deposited tailings are no longer in contact with oxygenated water, hence, metal leaching is less likely. Metals may have direct toxic effects on organisms or may bioaccumulate, with possible implications to the ecosystem or consumers.

2.2.3 Milling Reagents

Residual milling reagents are generally highly soluble and many are biodegradable, while others are not. Sodium cyanide is a reagent common to many operations. Other reagents vary largely between mills but typically include a list of organic and inorganic compounds including alcohols, amines, fuel oil, lime, metal salts, and sulfides (Table 2-1) (Read and Manser 1975, Davis et al. 1976, Read and Manser 1976, Down and Stocks 1977, USEPA 1988, Hesse and Reim 1993). Degradability, toxicity, and uptake potential are the factors most relevant to possible biological consequences.

Table 2-1

Predicted worst case milling reagent concentrations in the quartz Hill, Alaska, near-field plume (Poling et al. 1993).

Milling Reagent	Concentration (μg/L)
ALFOL6	1,003
Methyl isobutyl carbinol	1,9600
Fuel oil	1.0
Stepanfloat 85L	246
Dowfroth 250	67.0
Sodium Silicate	1,404

Nokes reagent	61.0
Resultant H_2S	49.0
M502	2,218
Lime	3,986
Aerodri	2.0

2.2.4 Sediment Deposition

The areas of deposition may alternate between active tailings deposition and natural sediment deposition due to meandering of the turbidity current. In the vicinity of the turbidity current, benthos obliteration is rapid and complete. With increasing distance from the turbidity current, deposition by settling fine tailings gradually tapers to zero. If tailings are resuspended, deposition may occur at all depths including the intertidal region. Sediment deposition may smother organisms or alter their habitat, forcing relocation.

3.0 RELEVANT BIOTA

Resident fauna could encounter combinations of suspended sediment, dissolved or sediment associated metals, milling reagents, and altered or obliterated spawning and foraging substrate. The consequences to an organism depend largely on species, life stage, mobility, depth distribution, and required spawning, foraging, or refuge habitats. Accordingly, the following descriptions will concentrate on these life history attributes, with the exception of mobility which is usually self evident. Biota at risk from STD include fish, zooplankton, benthic invertebrates, phytoplankton, and microbiota (Figure 2-1). The assemblages that could potentially be impacted by STD vary widely on a global scale, therefore, this section will focus mainly on species indigenous to coastal British Columbia and the southern coast of Alaska, though, even at this scale, assemblages of organisms are very site specific. The general distributional and life history traits detailed below should be somewhat applicable to other regions. This should not be considered a comprehensive listing of relevant species for the northwest coast of North America. Rather, it is a sampling of organisms covering a range of roles and habitats in the near shore assemblage.

3.1 Fish

Six species of Pacific salmon (*Oncorhynchus* spp.) reproduce in rivers along the northwest coast of North America (Groot and Margolis 1991). The fry of some species migrate to estuaries soon after emergence, while others spend one to two years in freshwater before migrating to the sea. All reside in saltwater during their juvenile and adult life stages. During their early life stages, estuaries can be important rearing areas (Healey 1982, Simenstad et al. 1982). After saltwater migration and maturation, salmon return to spawn in their natal streams which they detect, in part, through olfactory and visual cues. Generally, the near shore distribution of salmon appears to be close to the surface (Groot and Margolis 1991), though few detailed depth distribution studies have been performed. Particularly lacking are data pertaining to depth distributions of early life stage salmon. Ruggerone et al. (1990) tracked adult steelhead trout (*O. mykiss*) in two British Columbia channels and reported the fish as residing almost exclusively in the upper 5 m.

Quinn (1990) obtained similar results for adult steelhead trout, but found chinook salmon (*O. tshawytscha*) to swim somewhat deeper and sockeye salmon (*O. nerka*) to occasionally travel at 30 m, possibly for purposes of home stream orientation (Johnsen 1987). Pacific herring (*Clupea harengus pallasi*) exhibit a preference for spawning on seaweed in shallow water, in contrast to Atlantic herring which spawn at depth on rock or gravel (Saville and Wood 1977, DeGroot 1980). Both species are often found near the bottom (Horwood and Cushing 1977, Carlson 1980). Many stocks of herring undertake considerable migrations from their natal waters, though there are exceptions. Haldorson and Collie (1991) found larval Pacific herring to remain within the Alaskan sound of their origin. Similarly, Carlson (1980) reported a stock of adult Pacific herring (*C. harengus pallasi*) in southeast Alaska which did not make a long distance migration and held at depths of 85 m for extended periods.

The diversity of coastal fish species is immense and includes many overlapping and contrasting life histories. Some implications for herring larvae may also be applied to flounder (*Atheresthes* spp.), flathead sole (*Hippoglossoides elassodon*) and pollock (*Theragra chalcogramma*) due to observed co-occurrence of larvae in plankton samples (Messieh et al. 1981, Smith et al. 1991). The latter two species were the most abundant in a study of a southeast Alaska bay in which 28+ species of larval fish were collected (Haldorson et al. 1990).

Areas which are rocky or have high relief are preferred habitats for many species of fish, including juvenile and/or adult halibut (*Hippoglossus stenolepis*), lingcod (*Ophiodon elongatus*), and rockfish (*Sebastes* spp.) (Rosenthal et al. 1982, Krieger 1993, Bishop et al. 1993, O'Connell 1993, Yoklavich et al. 1993). Specifically, O'Connell and Carlile (1993) found boulder areas at greater than 108 m depth to be the most densely populated habitat for adult yelloweye rockfish (*Sebastes ruberrimus*). Generally, flatfish prefer sand to muddy-sand bottoms (Norcross and Muter 1993). As a whole, coastal fish feed on a variety of benthic and water column organisms and are found at all depths, sometimes exceeding 350 m.

3.2 Zooplankton

Zooplankton include the previously mentioned larval forms of fish, benthic invertebrates, and eggs of some species (meroplankton), as well as animals which spend their entire life cycle drifting with the currents (holoplankton). The vast majority of marine fish and benthic invertebrates undergo an early life stage planktonic period (Barnes 1987). In pelagic water, over continental shelves, the zooplankton assemblage has a higher proportion of meroplanktonic species than in oceanic waters (Raymont 1983). Zooplankton undergo diel vertical migrations which may be slight or up to 1000 m depending on the species and location (Parsons 1980, Raymont 1983). In Cook Inlet, Alaska, the zooplankton assemblage included three species of commercially important crabs, and larval forms of several shrimp species (Science Applications, Inc. 1977). Copepods, a class of crustaceans, were the most abundant taxa year round, typical of marine plankton assemblages. Copepods are also a common component of the benthos. Since most copepods feed on phytoplankton, and many marine animals, in turn, feed on copepods, they are a major trophic link in marine ecosystems (Barnes 1987).

3.3 Benthos

Aside from commercially important species, life histories of the deep benthos remain largely a mystery due to inherent difficulties in sampling and observation. Benthic invertebrates can be divided into two classes: the epifauna which live attached to, just above, or on the surface of the substrate, and the infauna which live within or partially within the sediment.

Many of the flatfish species discussed above are classified as epifauna, as are many crab, starfish, snails, scallops, and prawns. Sessile species include sea anenomes and mussels. Prawns (*Pandalus platyceros*) were found to prefer rocky substrates and depths of 70-100 m (Waldichuck and Buchanan 1980), whereas mating aggregations of tanner crabs (*Chionoecetes bairdi*) were observed at 150 m near Kodiak Island, Alaska (Stevens et al. 1993). The eggs of snow crabs (*Chionoecetes opilio*) are laid as deep as 350 m with adult males occurring down to 400 m (Adams 1979). Golden king crab (*Lithodes aequispina*) were found between 51 and 569 m in British Columbia fjords (Sloan 1985). Red king crab (*Paralithodes camtschatica*) were found to reside mainly between 50 and 150 m, with some occurrence at still deeper locations and were most abundant on sand and mud (Shirley et al. 1993). In contrast, higher densities of juvenile blue king crab (*Paralithodes platypus*) were found residing on cobble and gravel (Armstrong et al. 1985). Grain size is important in determining marine benthos distribution (Levinton 1982). In an Alaskan fjord formerly being considered for STD, a soft-bottom benthic survey revealed sediment grain size to be the primary factor separating distinct benthic assemblages (Cimberg 1982).

Infaunal organisms include burrowing crustaceans and bivalves, unsegmented worms (i.e., nematodes), and segmented worms (i.e., polychaetes). Burrowing infauna influence the exchange rate of ions and compounds across the sediment-water interface, affect the redox level, pH, and oxygen gradients, transfer reduced compounds into the surface sediment pore waters, and affect the cycling of carbon, nitrogen, sulfur, and phosphorus (Myers 1977, Levinton 1982).

Some benthic species are deposit feeders, ingesting large quantities of sediment and assimilating what digestible material is available. These animals include many gastropods, some bivalves, some polychaetes, annelids, and echinoderms. Other species, including many bivalves, polychaetes, and crustaceans, are suspension feeders, funneling or filtering material from the water column (Mann 1980), while some switch between the two modes of feeding (Lopez and Levinton 1987). Paul and Feder (1976) reported maximum depth occurrence of a soft-shell clam (*Mya priapus*) at 55 m. In a southeast Alaska survey down to 150 m depth, Woodby and Smiley (1993) found abundant giant red sea cucumber (*Parastichopus californicus*) on hard substrate with organic debris. Typically, polychaetes and bivalves are common residents of deep water, soft bottom assemblages (Sanders 1968).

3.4 Phytoplankton

Marine primary production is a measure of the energy available to primary consumers (i.e., zooplankton) through the conversion of light energy by phytoplankton. Photosynthetic phytoplankton are comprised mainly of diatoms and flagellates. The maximum depth at which light is sufficient to permit photosynthesis in the oceans can be as great as 200 m, but usually does not exceed 30 m (Fogg 1980, Barnes 1987). In productive waters, the presence of plankton reduces the euphotic/aphotic boundary considerably, however, non-productive settling phytoplankton are present to varying degrees at all depths (Round 1981).

During spring blooms at Alaska latitudes, the majority of phytoplankton settle to the bottom unconsumed where they serve as a food source for benthos (Graf et al. 1984, Fleeger et al. 1989, Fleeger and Shirley 1990). Phytoplankton assemblages are often dominated by one or two species, though many species will usually be present. Thirty species were identified in a Cook Inlet, Alaska survey, in which maximum photic zone depths were as shallow as 1 m and diatoms were dominant owing to the high silicate content of the waters (Science Applications, Inc. 1977).

3.5 Microbiota

Marine microbiota include bacteria, cyanobacteria, single cell algae, and protozoans. They may be free floating or attached to particles. Microbiota form much of the base of the benthic food chain, with some being more efficiently digested than others, depending on the grazer (Lopez and Levinton 1987, Decho and Castenholz 1986). Bacteria serve the additional role of mineralizing detritus and releasing nutrients for re-use by the primary producers (Blackburn 1987, Meadows and Anderson 1968). Bacteria are capable of degrading most naturally produced, and synthetic organic compounds (Pomeroy 1980). Many forms of microbiota form colonies on the surface of sand grains (Meadows and Anderson 1968). Their abundance increases with decreasing grain size due to higher surface to volume ratio of the sediment, and the growth of digestible species is stimulated by burrowing infauna (Hylleberg 1975, Levinton 1982). Bacteria secrete exopolymers which contribute to sediment stability, benthic larval settlement, and sequestering of organic matter and metals (Decho 1990).

4.0 DESCRIPTION OF CONSEQUENCES

The potential consequences of STD to biota include acute, chronic, and behavioral effects, habitat alteration, and bioaccumulation, the latter of which is not an effect, in and of itself, but a change. Secondary, or indirect, effects are usually chronic in nature. Experiments or surveys addressing the consequences of STD would most likely focus on effects of suspended sediment, metals, or milling reagents in the water column, exposure to deposited tailings, recovery after smothering, or effects of habitat alteration.

Metals or reagents may be dissolved in the water, in water overlying sediment, or in sediment pore water. Toxicants may also be bound to sediment and mobilized in the acidic digestive tracts of organisms upon ingestion. Uptake of dissolved toxicants may occur through the body wall, gills, mouth lining, or gastrointestinal tract. Adsorption to the integument of an animal may occur, possibly interfering with epithelial functions. Exposure may also result from ingestion of organisms which have accumulated or adsorbed a toxicant (Spacie and Hamelink 1985).

4.1 Acute Effects

Acute effects are defined as deleterious impacts caused by short-term exposure to a substance (Parrish 1985). Commonly this refers to the calculated concentration of a substance causing 50% mortality in 96 hours (96 h LC_{50}) based on a range of measured exposure concentrations. The definition of short-term is arbitrary, however, and the chosen end point need not be mortality. If a different effect is chosen as an end point, then this effect is described and the result presented is the EC_{50}, or the estimated effect concentration eliciting the described response in 50% of the experimental organisms after a given exposure duration.

4.2 Chronic and Secondary Effects

Experiments assessing chronic toxicity encompass the entire life cycle of an organism and, as a result, are generally of longer duration (Petrocelli 1985). Physiological impairment or genetic damage are the broad underlying causes for the effects typically investigated, these being, impaired fecundity, hatching, and growth. Commonly reported concentrations are the maximum acceptable toxicant concentration (MATC), the no observed effect concentration (NOEC), and the lowest observed effect concentration (LOEC). The MATC is hypothetical and is intermediate between the latter two which are actual measured exposure concentrations. Chronic effects need not result directly from exposure to a substance. For example, a fish species may be insensitive to a chemical, but growth may be reduced due to decreased density of a prey item which was directly affected. This is an example of a secondary, or indirect, chronic effect.

4.3 Bioaccumulation

Bioaccumulation occurs if the rate of uptake of a substance directly from water or sediment, or through consumption of food containing the substance, exceeds the rate of depuration. It is dependent on chemical concentration, speciation, exposure duration and pathway, and the ability of an organism to regulate the internal concentration of a chemical (Spacie and Hamelink 1985, Waldichuk 1985, Alliot and Frenet-Piron 1990).

Bioaccumulation data are reported as weight of a substance per unit of tissue (i.e., mg/kg). Whole body residues may be reported, or tissue may be isolated, such as muscle or liver. Often, a bioconcentration factor (BCF) is reported for a given exposure scenario. This number is calculated by dividing the concentration of a substance in an organism by the concentration in the water. Typically, bioaccumulation will reach a steady state, after which, investigators often transfer the experimental organism to clean water to determine the rate and degree of depuration (Spacie and Hamelink 1985). Determination of bioaccumulation is essentially meaningless if relation to effects on the organism or ecosystem, or to human consumption, cannot be made.

Biomagnification is the term given to increasing tissue concentration of a substance as it is transferred to higher trophic levels. It is the same as bioaccumulation, except it involves measuring tissue concentrations of at least two taxa linked by trophic interaction and is the sum of the amount of a substance accumulated from its surroundings plus the amount gained from its prey. Biomagnification is dependent on the chemical, the efficiency of transfer between trophic levels, and the structure of the food web. Closed food webs have an increased likelihood of biomagnification, whereas a flux of organisms into and out of an exposed assemblage has a diluting effect (Spacie and Hamelink 1985).

4.4 Behavioral Effects

Behavioral changes resulting from exposure to a substance are commonly in reference to avoidance, feeding rate, locomotor activity, learned behavior, and social behavior (Rand 1985). The relationship of these behavioral changes to an organisms ability to reproduce, grow, or survive are necessary if ecological significance is to be determined.

4.5 Habitat Alteration

Substrate type is critical to many benthic and epibenthic species as exemplified in Section 3 of this report. Many species require substrate grain size within limited bounds for foraging, refuge, and reproduction. In fine grained areas, natural sedimentation will cover anthropogenic fine grained deposits. Coarse grained or rocky areas may be permanently buried by such deposits, drastically altering the resident biotic community.

5.0 CONSEQUENCES TO BIOTA

The literature review for this report uncovered few peer-reviewed published articles addressing the consequences of STD to biota (Table 5-1). The following section will summarize physical and chemical data from STD operations relevant to discussion of the biological consequences, followed by a review of published and some unpublished investigations of the effects on biota. This will be followed by a review of research which indirectly pertains to STD. Information has also been compiled through comprehensive monitoring programs which have been employed at several STD operations (Evans et al. 1973, Pelletier 1982, Ellis 1987, Ellis 1988). Much of these data are not widely available and were not obtained for this review.

5.1 Black Angel Mine, Greenland

The Black Angel lead and zinc mine in western Greenland discharged tailings into the seasonally stratified Agfardlikavsa Fjord from 1973 to 1990 (Figure 5-1). The average tailing particle diameter was approximately 74 μm. They were composed of 50% marble and dolomite and 50% pyrite. The tailings were discharged at a depth of approximately 33 m at a rate of 500,000 metric tons/yr. The receiving fjord is 4 km long and has a maximum depth of 80 m. A 24 m deep sill separates the Agfardlikavsa Fjord from Quamarujuk Fjord, the latter of which received finer portions of the tailings from the former (Loring and Asmund 1989, Ellis et al. in press).

During operation, tailings, and possibly to a greater extent, shoreline waste rock, resulted in considerable increases of Zn and Pb, and to a lesser degree, Cu, Cd, As, Ag, and Hg, in the water (1 mg/L Zn and Pb), suspended matter (11.6 g/kg Zn and 5.9 g/kg Pb), and sediments (6.8 g/kg Zn and 2.5 g/kg) (Loring and Asmund 1989, Johansen et al. 1991). The fjord water often was nearly saturated with oxygen, enhancing the rate of metal dissolution from the tailings (Asmund 1975). Metal levels increased with depth in the summer and autumn and were mixed throughout the water column in winter and spring (Asmund 1986). The discharge contained xanthate, methyl isobutyl carbinol, sodium cyanide, copper sulfate, lime, and zinc sulfate (Ellis et al., in press).

Contamination of the fjord was reflected in elevated metal levels in new growth of seaweed (*Fucus disticus*), soft parts of mussel (*Mytilus edulis*) (Loring and Asmund 1989, Asmund et al. 1991), muscle of capelin (*Mallotus villosus*), liver and bone of shorthorn sculpin (*Acanthocottus scorpius*) and spotted wolf-fish (*Anarhichas minor*), and whole prawns (*Pandalus borealis*) (Johansen et al. 1991). Halibut (*Reinhardtius hippoglossoides*) and cod (*Gadus morrhua*) from the fjord did not have increased tissue metal concentrations. Mussels contained up to 5x (700 mg/kg) and 271x (1.3 g/kg) normal background Zn and Pb tissue levels, respectively. Prawn meat contained maximum concentrations of 47.2 mg/kg Zn and 1.42 mg/kg Pb. In comparison,

Table 5-1

Summary of peer-reviewed journal articles containing original biological data pertaining to modern submarine tailings disposal (STD).

STD Operation in Reference to	Description of Study	Reference
Black Angel, Greenland	Metal content; survey of fish, invertebrates, and macrophytes	Johansen et al. 1991
	Metal content; survey of mussels and macrophytes	Loring & Asmund 1989
	Metal content; survey of mollusks and macrophytes	Asmund et al. 1991
Island Copper, British Columbia	Effect of tailings deposition on an indicator organism	Jones & Ellis 1976
	Benthos observations relative to deposited tailings	Ellis & Heim 1985
	Benthic recolonization of tailings	Ellis & Hoover 1990b
AMAX/Kitsault, British Columbia	Zooplankton assemblages relative to suspended tailings	Mackas & Anderson 1986
	Laboratory, acute and sublethal effects of suspended tailings on zooplankton	Anderson & Mackas 1986
	Mesocosms, effects of settling tailings on plankton production and zooplankton assemblages	Parsons et al. 1986b
	Benthic recolonization of tailings	Ellis & Hoover 1990b
Quartz Hill, Alaska	Laboratory, acute effects of deposited tailings on fish, zooplankton, and benthos	Mitchell et al. 1985
	Laboratory, acute effects of two reactive metals on mussels	Morgan et al. 1986

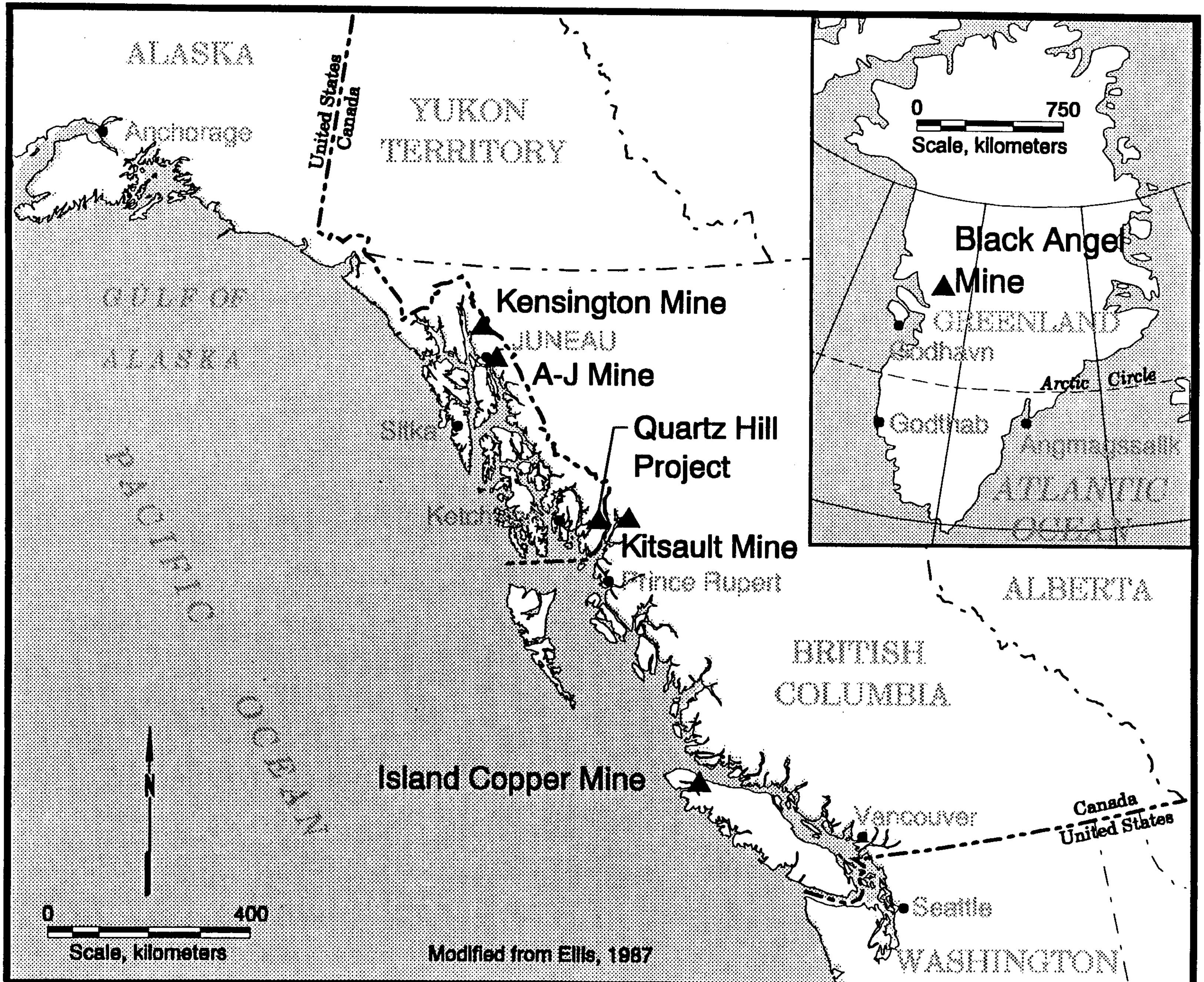

Figure 5.1 - Locations of sites discussed concerning biological consequences of submarine tailings disposal.

the Canada Food and Drug Act limited maximum Zn concentrations to 100 mg/kg and Pb to 10 mg/kg in marketable seafoods (Pelletier 1982). At 6 km from the discharge, mussel Zn and Pb levels were still 3.4x and 69x normal background, respectively. In subsequent mussel samples, Zn and Pb levels had increased by 39% and 60%, respectively, over the previous elevated samples, despite declines in metal concentrations of water and sediment. This indicated that concentrations in tissues had not yet reached steady state. Metals associated with suspended sediment were cited as the source for increased metal concentrations in mussel tissue whereas dissolved metals resulted in seaweed accumulation.

5.2 Island Copper Mine, British Columbia

The most intensively studied STD operation to date is the Island Copper Mine. The mine is located on Vancouver Island, British Columbia (Figure 5-1), and has been discharging tailings at a depth of 40 m into Rupert Inlet since 1971 at rates varying between 33,000 and 55,000 metric tons/d. Rupert Inlet is a well mixed fjord, 10 km long, approximately 1.8 km wide, and is connected to Holberg Inlet. Both fjords are separated from the Pacific Ocean by a 18 m deep sill. Holberg Inlet receives some dispersing tailings from Rupert Inlet. Prior to tailings disposal, the maximum depth of Rupert Inlet was approximately 165 m. The tailings are largely silt sized aluminosilicates. Milling reagents include a xanthate collector, polypropylene glycol, sodium cyanide, sodium hydrosulfide, sulphosuccinic acid, polyacrylamide, lime, and others (Poling et al. 1993).

In a laboratory study, upon submersion of tailings in sea water, Fe, Cu, and Pb water concentrations peaked after three hours and declined to background values within one month. Mg and Ni slowly increased during this time while Cd and Zn did not (Hoff et al. 1982). The advancing tailings deposit could be delineated by higher acid-leachable Cu and Zn relative to natural sediment (Thompson and Paton 1975). Copper flux to overlying sea water was similar to that of natural sediment and probably a result of decomposing organic material and oxidation of sulfide minerals (Pedersen 1985). Molybdenum flux was appreciable, but of little concern due to the high flushing rate of the basin and high natural concentrations of Mo in sea water (Drinkwater and Osborn 1975, Pedersen and Losher 1988). One of the main reasons cited by Pedersen and Losher (1988) for a lack of appreciable metal leaching into Rupert Inlet was the rapid burial of the tailings by more tailings, limiting their exposure to oxygenated bottom waters. The authors questioned if mobilization of metals may increase when the operation ceases but before a natural sediment layer covers the deposit.

There was a depletion of interstitial phosphate in the tailings deposit due to lime additions (Pedersen 1984, Pedersen and Losher 1988). The investigators concluded that phosphate depletion was not a problem in Rupert Inlet due to rapid water renewal, but in shallow water, with less water exchange, such depletion could result in reduced biological production.

Occasionally, tidal currents flowing into Rupert Inlet through a narrow passage have caused a considerable resuspension of deposited tailings. The possibility of this occurrence was apparently disregarded during planning studies. The suspended tailings have extended throughout Rupert Inlet and into adjoining water bodies, periodically rising to the water surface and depositing in intertidal areas. In the process, a formerly rocky intertidal area has been converted to a fine grained habitat (Poling 1992). No information was found addressing the biological relevance of this habitat alteration. Waldichuck and Buchanan (1980) mention the possibility for effects on local fauna including prawn (*Pandalus platyceros*), dungeness crab (*Cancer magister*), salmon (*Oncorhynchus* spp.), herring (*Clupea* sp.), large mussel (*Mytilus californianus*), various bottom fish species, and seaweed (*Fucus* spp.).

Goyette and Nelson (1977) commented that adult and juvenile salmon must pass through areas affected by tailings resuspension. Ellis and Heim (1985) observed layers of suspended sediments in the upper 5 m of water, increasing with depth from 12 m to the bottom. They also noted an absence of zooplankton in the layers with visible suspended sediment. Ellis (1989) stated that phytoplankton crops have not dropped since the Island Copper operation commenced. A study employing acoustic sounders in Rupert Inlet detected what appeared to be fish in close association with the turbidity current at approximately 130 m (Hay et al. 1982).

The extent of benthic smothering and recolonization has been investigated through sampling and a submersible survey. The purpose of the submersible survey was to quantify the benthos between 13 and 56 m depth at three sites within the tailings impacted area and one reference site (Ellis and Heim 1985). At depths of 13-16 m in approximately 5 cm of accumulated tailings, burrowing sea anemones (*Pachycerianthus fimbriatus*) dominated the epifauna. At 16-31 m with a tailings layer of approximately 0.5 cm, starfish, fish, and clams were present in less abundance than sea anemones at the previous location. At 27-56 m depth, a heavy tailings deposit was present due to resuspension and very few epifauna were seen. In contrast, the reference station (42-51 m) supported a relatively abundant and diverse assemblage of epifauna typical of shallow British Columbia fjords, including starfish, several fish species, holothuria, and burrowing anemones.

Benthic sampling suggested a complete absence of fauna only in areas directly affected by the erosion from the turbidity current (Ellis and Hoover 1990b). Mobile polychaetes were found to dominate benthic assemblages in areas most recently smothered, followed in succession by sedentary polychaetes. Sites with more advanced recovery had increased crustacea, mollusk, and echinoderm abundance. Taylor (1986) studied invertebrate colonization of *in situ* trays containing Island Copper tailings in Rupert Inlet. Number of individuals and taxa increased through the 12 month study, at which time, an equilibrium had not been reached. Polychaetes, settling as larvae, dominated the fauna. The polychaete, *Ammotrypane aulogaster*, may be a valuable indicator species of reduced benthic productivity resulting from subobliterative tailings deposition in Rupert Inlet (Jones and Ellis 1976).

Young and Ellis (1982) concluded that there was no evidence of bioaccumulation of metals by biota. Waldichuck and Buchanan (1980) came to the same conclusion, but stated that long-term changes in the tailings deposit, through the addition and decomposition of organic material, could not be predicted. Poling et al. (1993) cite the deposition of organic detritus as promoting oxygen depletion with a resulting stabilization of metal compounds.

Poling et al. (1993) briefly summarized an extensive biological monitoring program surrounding the Island Copper operation which included bioassays, species distribution, and metal content surveys. They reported lower benthos diversity and abundance in tailings depositional areas, but no consistent reduction in crab catch associated with tailings discharge. They also found extensive benthic repopulation in areas which had not undergone tailings deposition for 12 months, though assemblages of recolonizing organisms differed from reference sites. Monitoring also revealed no large scale affects on average chlorophyll *a* concentration or zooplankton populations (Poling 1979).

5.3 AMAX/Kitsault Mine, British Columbia

From April 1981 to November 1982, tailings from the AMAX/Kitsault molybdenum mine were discharged at 50 m depth into Alice Arm, a British Columbia fjord (Figure 5-1). Alice Arm is a seasonally stratified fjord. It has a maximum depth of 386 m, is 19 km long, 2 km wide, and has a 20 m deep sill at its mouth. Dissolved oxygen throughout the water column is typically between 60 and 70%. Twenty percent of the tailings from the mine were <50 μm in size (Poling et al. 1993).

Goyette et al. (1985) reported a mid-water tailings plume between 80 and 120 m within which the following maximum particulate metal concentrations were measured:

Cu = 3.0-25.1 μg/L
Pb = 100-314 μg/L
Zn = 167-539 μg/L
Cd = 4.0-10.7 μg/L
Mo = 30.0-130.0 μg/L

In the tailings deposit, sediment metal concentrations were:

Cu = 123 mg/kg
Pb = 435 mg/kg
Zn = 561 mg/kg
Cd = 14.3 mg/kg
Mo = 291 mg/kg

Two peaks in sediment metal concentrations were observed, presumably the peak concentration nearer to the sediment surface resulted from the 1981-1982 operation, whereas the peak concentration found deeper in the sediment was the result of a riverine tailings disposal operation which ceased in 1972. Pedersen and Losher (1988) found the deposited tailings to be releasing considerable quantities of Mo, though the resultant contribution to the fjord waters was considered minor.

In laboratory studies, the ability of benthos to survive continuous AMAX/Kitsault tailings deposition was determined (Reid and Baumann 1984). Benthos, listed in order of decreasing tolerance, were four subtidal bivalve species, polychaetes (*Sternaspis scutata*), juvenile tanner crabs, and three species of intertidal bivalves. The instantaneous burial depths at which animals could migrate through were >30 cm and 20-30 cm for subtidal bivalves and *S. scutata*, respectively.

Kathman et al. (1983) traced benthic smothering resulting from the AMAX/Kitsault operation and found few species up to 10 km from the discharge, coincident with the reported front of the tailings deposit. Eleven months after cessation of discharge, patterns of abundance of benthos colonizing the tailings deposit could no longer be differentiated from reference sites. Polychaetes, mollusks, and crustaceans colonized the disturbed areas in equal proportions (Kathman et al. 1984).

Ellis and Hoover (1990b) drew similar conclusions for the AMAX/Kitsault mine as were reported above for the Island Copper mine with regard to the general nature of benthic recovery, though there was little overlap between the two sites in assemblages at the species level. Three to four years after closure, benthic diversity had increased, but was not considered as diverse as pre-impact, in contrast to the findings of Kathman et al. (1984).

Published biological reports on the effects of suspended sediment from the mine include a laboratory investigation, a field experiment, and a field survey. In the laboratory study, bioassays were performed with two copepod species (*Calanus marshallae* and *Metridia pacifica*) and a euphausiid (*Euphausia pacifica*) (Anderson and Mackas 1986). All three species ingested tailings. The lowest concentration of suspended tailings to produce a clear decrease in survival was 560 mg/L in *Euphausia pacifica*. Behavior was unaffected at 100 mg/L in *C. marshallae*, and oxygen consumption was depressed, though the relevance of this was unclear. In comparison, the concentration of suspended sediment in the mid-water plume was $\leq$15 mg/L.

The field experiment was three weeks in duration, utilizing large *in situ* enclosures (Parsons et al. 1986). Two of the enclosures received additions of AMAX/Kitsault tailings suspensions of 39 and 297 ppm (presumably mg/L). The greatest impact was on the phytoplankton in the 0-5 m layer of both treatment concentrations where a delay in primary production during a bloom was noted. Also, in the high concentration enclosure, there was a shift in the primary producer assemblage resulting in proportionally fewer diatoms and more flagellates. Zooplankton community structure was unaffected and production increased, in contrast to primary production. In the field survey, concentrations of suspended tailings were reported to be 2-100 mg/L, bracketing the low concentration in the above study (Parsons et al. 1986). Normal background levels were $\leq$1.0 mg/L (Mackas and Anderson 1986). The authors noted minor changes in zooplankton community structure, possibly as a result of the tailings discharge.

Brief mention was given to bioaccumulation resulting from the AMAX/Kitsault tailings deposit, most notably were a 3x increase in Pb and 1.8x increase in Zn in tissues of bivalves (*Yoldia traciaeformis/montereyensis*) within six months after discharge initiation (Goyette and Christie 1982b). Other species sampled showed no significant increase in tissue metal levels. A more detailed unpublished report from which these conclusions were drawn was not obtained for this review (Goyette and Christie 1982a).

5.4 Proposed STD Sites

The proposed Quartz Hill molybdenum mine near Ketchikan, Alaska (Figure 5-1) was the first project to attempt permitting for modern STD in the U.S.. The EPA denied the permit application in 1990. Considerable research was performed in preparing the environmental impact statements for this project (USDA 1988a, USDA 1988b, USEPA 1988), though few study results were published.

Hesse and Reim (1993) reported the maximum euphotic zone as 25-30 m in the fjords proposed for STD, comparable to the findings of Burrell (1981) who conducted an in depth investigation of the physical and geochemical aspects of the fjords. Projected tailings deposit depths over the 55 year predicted duration of the operation were approximately 100 m in one fjord option to 150 m in the other. Covered benthic surface areas would have ranged from 16.5 to 20.5 km^2. Computer modeling predicted 5-20 mg/L suspended sediment at depths of 50 m, though short-term surface water impacts were beyond the capability of the model. A report by the USDA (1981) cited the possibility of tailings resuspension during certain periods of the year, stating that changes in circulation due to partial filling of the discharge basin with tailings and resulting effects on tailings deposition could not be predicted.

Poling et al. (1993) listed the predicted milling reagent concentrations in the near field plume. Of the reagents listed in Table 2-1, ALFOL6 and methyl isobutyl carbinol are biodegradable and exhibit low toxicity (USEPA 1988). The remaining reagents are either non-biodegradable, or their biodegrability is not known. Fuel oil, while a minor constituent of the predicted effluent, is highly toxic, as is H_2S which results from the reaction of Nokes reagent and water (USEPA 1988).

The USDA (1981) reported phytoplankton to be the major type of plant material in the fjords, dominated by diatoms and dinoflagellates. Primary production was limited to depths above 25 m (USEPA 1988). In late summer samples, the zooplankton assemblage was dominated by calanoid copepods, followed by cnidarians and amphipods (USDA 1981). Other abundant taxa included crangonid shrimp larvae, cladocerans, mollusk larvae, and bryozoans. Dominant deep water benthos were echinoderms, polychaetes, clams, and amphipods. Commercially important species included shrimp, dungeness and tanner crab, flatfish, salmon, and herring (Cimberg 1982, USEPA 1988, Hesse and Reim 1993). One of the largest herring spawning and rearing grounds in southeast Alaska is located in the proposed discharge fjords (USEPA 1988).

The USDA (1981) predicted that much of the deep benthos would likely have been completely obliterated soon after discharge commenced, and possibly some rocky and coarser grained substrates would have been covered by tailings. With regard to recolonization, Burrell (1981) described *in situ* experiments that were performed with pilot tailings. Trays containing tailings were placed on the bottom of the proposed STD fjords. The trays were retrieved after 2 to 12 months and colonizing communities were to be quantified, however, no results were presented. Winiecki and Burrell (1985) presented results from nearly identical experiments in one of the proposed STD fjords using defaunated natural sediment rather than tailings. Conclusions were less relevant to STD but lent some insight into recolonization potential in the fjord. The investigators found an unequilibrated trend of increasing numbers of taxa, density, and biomass over the 78 week experiment, marked by seasonal variations.

Mitchell et al. (1985) published the results of laboratory acute bioassays performed to determine the effects of deposited tailings. Tailings were mixed with natural sediment over a range of ratios. Toxicity results were presented as a function of grams of tailings per liter of water. Fifty percent of coho salmon (*Oncorhynchus kisutch*), amphipods (*Rhepoxynius abronius*), euphausiids (*Euphausia pacifica*), and mussel larvae (*Mytilus edulis*) died within 96 h, 10 d, 96 h, and 48 h, respectively, in chambers containing settled natural sediment mixed with 100 to 200 g/L tailings in static water.

Toxicity decreased as the proportion of natural sediment increased, indicating that the tailings were the source of toxicity. Numerous dissolved metals were detected in the leachate water, notably, Mn at 1.750 mg/L and Mo at 600 µg/L after 5 d. Morgan et al. (1986) studied the individual and combined toxicity of Mn and Mo to *M. edulis* larvae. Results indicated that these two metals, in the absence of the other leachates from the previous experiment (Mitchell et al. 1985), were not the cause of the observed mortality, though uninvestigated combinations of other leachate components left open the possibility of a contribution of Mn and Mo to the toxicity of the tailings.

Close inspection of an unpublished report (E.V.S. 1984) written by the same investigators added additional information to the published findings (Mitchell et al. 1985). The following results were extracted from the appendices of the unpublished data report and represent the effects of static exposure to settled tailings with no natural sediment added. Mortality was 100% for amphipods (*Rhepoxynius abronius*) and crab (*Cancer magister*) larvae after $\leq$10 d. The percent of abnormal mussel larvae was 100% after $\leq$48 h. The percent of affected organisms at intermediate durations was not given, hence the $\leq$ symbols. The static exposures included an equal volume of liquid and solid tailings components. These experiments demonstrated that the elutriate from the tailings may have been acutely toxic. Flow-through or *in situ* exposures would have been the next step in determining toxicity of the tailings under more realistic conditions in which the elutriate would have dispersed, as it would in an actual STD discharge.

Bioaccumulation experiments were described in the same report. Analyzed metals were Cd, Cu, Fe, Pb, Mn, and Mo. The tailings were mixed with natural sediment to avoid acute toxicity. It appeared that clam (*Macoma balthica*), mussel (*Mytilus edulis*), juvenile flatfish (*Citharichthys stigmaeus*), and crab (*Cancer magister*) did not accumulate metals appreciably from tailings diluted with natural sediments (4:1 and 2:1, natural sediment:tailings), relative to control animals.

The USEPA (1988) summarized the potential impacts of various options for the Quartz Hill STD project. Acute toxicity of tailings was expected to be low. The potential for toxicity of metals in the liquid portion of the effluent was not known. Information on organic milling reagents was considered insufficient to determine potential toxicity, particularly from quaternary ammonium salt polymers. Bioaccumulation of metals by zooplanktivores, including herring, was considered a possibility. Biomagnification of metals was not expected. Reduction in juvenile rearing habitat was deemed possible due to avoidance of projected increases in suspended sediment in the upper mesopelagic zone. Juvenile salmonids were not expected to be affected by increased suspended sediment. Significant smothering of benthos was predicted. For one option, bathymetry changes were expected to exceed the depth ranges of some resident species, thereby permanently altering the benthic community. The time needed for recolonization was not known.

In addition to the Quartz Hill Project, two other projects in southeast Alaska have evaluated the potential for STD (Figure 5-1). One of these, the proposed A-J gold mine, south of Juneau, may represent the best opportunity for successful STD to date. Developers proposed discharging tailings at the head of Taku Inlet, an area of high, natural inorganic sedimentation in the form of glacial flour (USBLM 1991). North of Juneau, preliminary screening revealed possible suitability of STD for the proposed Kensington gold mine (Echo Bay Mgmt. Corp. 1988, Kessler/E.V.S. 1992). Natural sedimentation at this site is more comparable to the British Columbia operations. Local fauna are diverse, including scallops, mussel, limpet, herring, halibut, sablefish, flounder, cod, pollock, skate, rockfish, king crab, and tanner crab (Calvin 1977, Echo Bay Mgmt. Corp. 1988, USDA 1991).

5.5 Related Research

Published data pertaining to activities which cause similar perturbations to STD are useful, but not directly applicable. Some research, such as effects of barge dumping of waste, or dredging, could have similar consequences to STD including increased levels of suspended sediment, release of metals and other sediment associated contaminants, and smothering of benthos. The many differences, however, do not allow direct application of the data to STD operations.

Reasons for the limited usefulness of these data include the likely differences in contaminant types, submarine aging of dredged sediment, and possible acclimation of resident biota to associated contaminants. Point-source industrial wastewater discharge is likely to impact some similar assemblages of biota as found in an STD near-field zone, but the mixture of the effluent is unlikely to have any applicability to an STD effluent. Industrial effluents also do not usually include high levels of suspended sediment. These related activities do, however, allow for some general comparisons. Dredging research, for example, can be useful in predicting the effects of suspended sediment on primary production, and laboratory research pertaining to individual components of STD plumes, such as metals, can indicate sources of tailings toxicity.

5.5.1 Other Large Scale Marine Perturbations

In countries with less stringent environmental regulations, and prior to the advent of more strict environmental legislation in other countries, tailings were, or are, often dumped in intertidal waters or marine tributary rivers. Consequences of the following operations are not presented as cases of modern STD, but rather, to serve as extreme examples illustrating the potential consequences of poorly conceived marine tailings discharge.

Surveys of biota impacted by intertidal tailings dumping in Chile indicated severe reductions in benthos abundance and diversity in addition to reductions in plankton communities due to continual sediment suspension caused by waves and currents (Castilla and Nealler 1978). When the dump site was moved to a different intertidal area, massive mortalities of fish, mollusks, starfish, limpet, sea urchins, and crab were reported. Wong et al. (1978) surveyed the benthos associated with intertidal iron ore mine tailings in Hong Kong and noted a dominance of crustaceans and bivalves in tailings areas, with an absence of other organisms found in non-tailings areas.

Between 1972 and 1974, the Jordan River Mine discharged copper mill tailings to a wave exposed platform. Due to breaks in the outfall, a tailings beach formed. There was evidence of Cu accumulation by clams, followed by decreasing Cu concentrations over the next six years (Ellis 1983).

In a Papua New Guinea operation (Ok Tedi), the finer portion of tailings were discharged into a river which carried some of them to the sea (Baker et al. 1990). River sediment had Cu concentrations of 850 mg/L. Receiving seawater contained suspended sediment with up to 13.3 and 0.8 µg/L Cu and Cd, respectively. Sediment collected from the river delta revealed a possible, slight increase in Cu concentration resulting from the discharge (Alongi et al. 1991). Consequences to marine biota were not addressed, though an 80% reduction in a portion of the rivers fish population was reported.

Riverine tailings disposal has also been practiced at a copper mine at Britannia Beach, British Columbia. The river carried the tailings into a fjord where acid leachable Cu concentrations in submerged tailings ranged from 83 to 1394 µg/L and Zn concentrations ranged from 97 to 1324 μg/L (Thompson and McComas 1974). Copper, and less commonly, Zn, were elevated in the tissues of some suspension feeders and *Fucus* sp. (Ellis 1983).

The contribution of the Britannia Beach discharge to light attenuation and resulting reduced phytoplankton growth was found to be minor compared to natural riverine input (Stockner and Cliff 1977). Submersible surveys during the operation, however, indicated low abundance of benthos in areas of high suspended sediment resulting from the discharge (Levings and McDaniel 1973). Surveys of the fjord deposit indicated partial recovery of benthic abundance and diversity 12 years after closure of the operation (Ellis and Hoover 1990a). Fauna on the tailings beds were less diverse than non-tailings assemblages. Diversity increased with depth, as did the thickness of the overlying natural sediment which ranged from 1 to 25 cm. Bright and Ellis (1989) reported histological legions in clams (*Macoma carlottensis*) collected near the former discharge site, probably resulting from metal toxicity.

Dredging suspends sediment and creates conditions similar to the resuspension events which occur at the Island Copper Mine. Similarities can be drawn because the sediment is often contaminated, allowing investigation of combined physical and chemical effects. The sediment, however, has had more time to come into chemical equilibrium with the water, degradable contaminants are not likely present, and the benthos have had the opportunity to become acclimated or adapted to the contaminants, unlike STD scenarios.

A marine placer mine (the WestGold) operated between 1985 and 1990 at maximum depths of 21 m near Nome, Alaska. Tailings were discharged back to the sea bed. Red king crab (*Paralithodes camtschatica*) were monitored for metals and found to accumulate As and Ni, though high variability did not allow for firm conclusions as to the source of the metals. It was estimated that it would take 3-4 years for complete recolonization of crabs and mollusks on sand substrate, and five or more years on cobble (Poling et al. 1993).

Parrish et al. (1989) documented accumulation of metals from laboratory exposures of polychaetes (*Arenicola cristata*), pink shrimp (*Penaeus duorarum*), and oysters (*Crassostrea virginica*) to redeposited dredged sediment. After exposure to dredged sediment contaminated with As, Cd, Cr, Cu, Hg, Ni, Pb, Se, and Zn, polychaetes accumulated only Hg, oysters accumulated As, Cd, Cu, Hg, and Zn, and shrimp accumulated Cd, Cr, Hg, and Pb. Recolonization experiments were also performed, with no significant differences in diversity or abundance of recolonizing macrobenthos compared to reference sediment.

Limpsaichol and Pooptech (1984) found primary production to decline in response to dredging, particularly due to inhibition of smaller phytoplankton species. Suspended sediment was highest in the upper 5 m of water, below which, concentrations decreased until increasing again in a layer above the substrate.

Salo et al. (1979) performed *in situ* and laboratory bioassays with juvenile chum salmon (*Oncorhynchus keta*) to assess possible effects of a dredging operation on the Washington coast. In static bioassays, 96 h LC_{50} values ranged from 15.8 and 54.9 g/L and fish were more tolerant of sediment with a mean diameter of 44 μm than of 64 μm diameter sediment. Suspended sediment levels near the dredging operation ranged from 4 to 94 mg/L. *In situ* and laboratory mortalities were low and not attributed to the operation. Avoidance behavior was estimated to occur at 182 mg/L suspended sediment and was attributed to the sediment and not to associated contaminants.

Avoidance behavior of juvenile herring (*Clupea harengus harengus*) in response to dredged sediment was demonstrated by Johnston and Wildish (1981) at various light intensities. Fish avoided suspended sediment concentrations of 9 and 12 mg/L and the behavior was attributed, in part, to visual cues. Using similar sediment, Messieh et al. (1981) demonstrated decreased feeding in larval herring exposed to 3 mg/L suspended sediment.

Organisms that could encounter offshore mining perturbations are likely different than assemblages near STD operations because much of the mining takes place in less productive oceanic waters. As with STD, Hirota (1981) noted that subsurface discharge of solid waste below the photic zone and pycnocline minimized the effects of suspended sediment. At greater depth, the investigators suggested the possibility of reduced or impaired zooplankton assemblages due to lower nutritional content of available suspended matter and higher concentrations of metals, though this possibility was not actually investigated. Hu (1981) found zooplankton to increase deposition of offshore mine tailings through fecal pelletization, though the benefit was countered by leaching of trace elements from sinking feces. The same study reported uptake of metals by copepods (*Undinula darwinii*) through tailings ingestion. Sediment plume modeling of surface disposal predicted the possibility of some light reduction up to 100 km from a discharging ship, though the impact on phytoplankton was not investigated (Lavelle and Ozturgut 1981).

5.5.2 Suspended Sediment Research

Considerable research has been performed on the effects of suspended sediment, though much of this has work focused on effects on salmonids in streams (Crouse et al. 1981, Sigler et al. 1984, Berg and Northcote 1985, Redding et al. 1987). Table 5-2 summarizes much of the available data relating to the effects of suspended sediment on marine fish. Data pertaining to invertebrates were often in reference to contaminated sediment and, therefore, difficult to summarize in table form. Research pertaining to contaminated sediment is addressed elsewhere in this report. Effects of suspended sediment on phytoplankton are briefly addressed in the previous STD site summaries.

In reviewing Table 5-2, acute lethality of suspended sediment to fish was typically in the range of 1 to 10 g/L with some indication of increased larval sensitivity (Sherk et al. 1975). The most sensitive end-point was avoidance behavior (Messieh et al. 1981, Wildish and Power 1985). In a review of the effects of suspended sediment on juvenile salmonids (Servizi 1990), fish were found to spend more time swimming at the water surface at suspended sediment concentrations above 140 mg/L, and had reduced resistance to bacterial infection, though minimum concentrations causing the latter effect were not given. After the 1980 Mount St. Helens eruption, sub-yearling chinook salmon (*Oncorhynchus tshawytscha*) estuarine distribution changed in response to increased suspended sediment (Emmet et al. 1990). It was not known if this was direct avoidance or a response to effects on prey.

McFarland and Peddicord (1980) calculated the following 200 h LC_{50} values for 4.5 μm diameter kaolin: coast mussel (*Mytilus californianus*), 96 g/L, spot-tailed sand shrimp (*Crangon*

Table 5-2

Effects of suspended sediment on marine fish.

Organism	Duration/Effect	Conc (g/L)	Reference
Alosa sp.			
alewife, *A. pseudoharengus*	no effect, egg hatching	1	Auld & Schubel 1978
American shad, *A. sapidissima*	no effect, egg hatching	1	
larvae	72h, decreased survival	≥0.1	
blueback herring, *A. aestivalis*	no effect, egg hatching	1	
Agonus sp. armed bullhead, *A. cataphractus*	72h, 60% mortality	10	Blackman & Wilson 1973
Anchoa sp. bay anchovy, *A. mitchilli*	24h LC_{50}	4.71	Sherk et al. 1975
Atherinidae sp. Atlantic silversides, *A.* sp.	24h LC_{50}	2.50	
Brevoortia sp. menhaden juveniles, *B. tyrannus*	24h LC_{50}	2.47	
Clupea sp. herring, *C. harengus pallasi*	96h, decreased survival	4	Morgan & Levings 1989
larvae	no effect, egg survival and hatching	10	
C. harengus	no effect, egg hatching	7.55	
	decreased feeding avoidance	≥0.003 <0.012	Messieh et al. 1981
C. harengus pallasi	increased feeding	0.5-1	Boehlert & Morgan 1985
Cymatogaster sp. shiner perch, *C. aggregata*	100h LC_{50}	6	McFarland & Peddicord 1980
Fundulus sp. commom mummichog, *F. heteroclitus*	24h LC_{50}	39.0	Sherk et al. 1975

Table 5-2

Effects of suspended sediment on marine fish.

Organism	Duration/Effect	Conc (g/L)	Reference
striped killifish, *F. majalis*	24h LC_{50}	38.2	
Hypomesus sp. surf smelt, *H. pretiosus*	7d, decreased survival	4	Morgan & Levings 1989
larvae	decreased hatching	4	
Leiostomus sp. spot, *L. xanthrus*	24h LC_{50}	9.80	Sherk et al. 1975
Morone sp. white perch larvae, *M. americana*	decreased hatching 24h LC_{50} 48h LC_{50}	1 3.73 1.55	Auld & Schubel 1978
adults	24h LC_{50} 48h LC_{50}	9.85 2.96	Sherk et al. 1975
striped bass, *M. saxatilis*	decreased hatching	1	Auld & Schubel 1978
	decreased hatching	0.89	Hanson & Walton 1990
larvae	48-96h, decreased survival	≥0.5	Auld & Schubel 1978
	24h LC_{50} 48h LC_{50}	4.85 2.80	Sherk et al. 1975
	48h LC_{50} decreased feeding	6.29 0.20	Hanson & Walton 1990
Oncorhynchus sp. chum salmon juveniles, *O. keta*	96h LC_{50}	16-55	Salo et al. 1979
Oncorhynchus sp. chinook salmon juveniles, *O. tshawytscha*	increased foraging	0.05-0.20	Gregory 1990
Ophidion sp. lingcod, *O. elongatus*	10d, decreased survival	10	Morgan & Levings 1989
larvae	no effect, egg survival and hatching	10	

Table 5-2

Effects of suspended sediment on marine fish.

Organism	Duration/Effect	Conc (g/L)	Reference
Osmerus sp. smelt adults, *O. mordax*	avoidance	0.022	Wildish & Power 1985
Perca sp. yellow perch, *P. flavescens*	48-96h, decreased survival	≥0.5	Auld & Schubel 1978
larvae	no effect, egg hatching	1	

nigromaculata), 50 g/L, grass shrimp (*Palaemon macrodactylus*), >77 g/L, dungeness crab (*Cancer magister*), 32 g/L, and polychaete (*Neanthes succinea*), 48 g/L. One hundred hour LC_{50} values were as follows: tunicate (*Acidia ceratodes*), 38 g/L, and amphipod (*Anisogammarus confervicolus*), 78 g/L. The authors concluded that organisms restricted to muddy bottoms were more tolerant to high suspended sediment levels. Peddicord (1980) exposed sand shrimp (*Crangon* spp.) and coast mussels (*M. californianus*) to a mostly silt size (2-50 μm) suspended sediment. Approximately 30% of the sand shrimp died after 19 d exposure to 19.7 g/L suspended sediment. Approximately 40% of the coast mussels died from 19 d exposure to 15.5 g/L.

Sublethal concentrations of suspended sediment can change invertebrate feeding behavior. Polychaetes (*Boccardia pugettensis* and *Pseudopolydora kempi japonica*) switched from deposit feeding to suspension feeding as suspended sediment levels increased (Taghon and Greene 1992). Also, Kiorboe et al. (1980) found that *Mytilus edulis* is well adapted to efficient feeding at suspended sediment concentrations up to 55 mg/L through increased ingestion and particle selection.

Turk and Risk (1981) performed smothering experiments with amphipods (*Corophium volutator*), bivalves (*Macoma balthica*), and soft-shelled clams (*Mya arenaria*) *in situ* and in a laboratory. Sediment accumulation of 2.8 cm in 7 d significantly reduced the density of amphipods. Bivalves were unaffected at sedimentation rates up to 10.2 cm/mo due to the ability to burrow upward and elongate their surface feeding siphon. In instantaneous laboratory sediment additions, 50% of soft-shelled clams were killed by 3 cm deposition of silt, 6 cm of fine sand, or 24 cm of coarse sand. Using the same species buried with 7.5 cm of silty-sand, Glude (1954) observed 21% mortality, and decreasing tolerance with decreasing clam size. Brenchley (1981) found dense, shallow-dwelling, tube builders (a polychaete, *Rhynchospio arenicola* and a crustacean, *Leptochelia dubia*) to be more sensitive to sediment deposition than burrowing or mobile invertebrates.

5.5.3 Metals Research

Many metals are essential to survival but, as with non-essential metals, are toxic if present in high enough concentrations. For a metal to be toxic, it must be bioavailable. The toxicity of a metal varies greatly depending on its chemical state and the exposed organism. Generally, the dissolved inorganic fractions are the most toxic, and often, toxicity is due to impaired osmoregulatory ability (Lewis et al. 1972, Giles et al. 1984). Metal toxicity generally decreases with increasing salinity, organic content, and oxygen concentration, and decreasing temperature (Forstner and Wittmann 1979), though exceptions are common. With all of these contributing variables, metal toxicity data must be carefully scrutinized.

Tables 5-3, 5-4, and 5-5 summarize the toxicity of various metals to marine fish, invertebrates, and phytoplankton. Due to the unpredictability of metals toxicity, the lists are fairly exhaustive. Most of the data presented concern metals considered highly toxic and abundant, particularly Hg, Pb, and Cd, as opposed to less toxic metals such as Fe and Br, or toxic but rare or insoluble metals such as Ti, Ga, and Ba. Chronic toxicity data were presented whenever found, however, most of the data listed are in reference to acute toxicity. Though likely of limited STD relevance, these data still serve as a general ranking of metals toxicity and species sensitivity where

Table 5-3

Effects of Inorganic Metals on Marine Fish.

Metal	Organism	Duration/Effect	Conc (mg/L)	Reference
As	dab adults, *Limanda limanda*	96h LC_{50}	28.5	Taylor et al. 1985
	mullet adults, *Chelon labrosus*	96h LC_{50}	27.3	
	pink salmon, *Oncorhynchus gorbuscha*	96h LC_{50}	3.8	Eisler 1988a
B	dab adults, *Limanda limanda*	96h LC_{50}	74.0	Taylor et al. 1985
Ba	sheepshead minnow juveniles, *Cyprinodon variegatus*	96h LC_{50}	>500	Heitmuller et al. 1981
Cd	hardyhead pre-adults, *Atherinasoma microstoma*	168h LC_{50}	16.6	Negilski 1976
	mullet juveniles, *Aldrichetta forsteri*	120h LC_{50}	14.3	
	mummichog adults, *Fundulus heteroclitus*	96h LC_{50}	22.0	Eisler & Hennekey 1977
Cr	dab adults, *Limanda limanda*	96h LC_{50}	28.5	Taylor et al. 1985
	hardyhead pre-adults *Atherinasoma microstoma*	168h LC_{50}	31.6	Negilski 1976
	mullet adults, *Chelon labrosus*	96h LC_{50}	27.3	Taylor et al. 1985
	mullet juveniles, *Aldrichetta forsteri*	96h LC_{50}	31.2	Negilski 1976
Cu	dab adults, *Limanda limanda*	96h LC_{50}	0.3	Taylor et al. 1985
	mullet adults, *Chelon labrosus*	96h LC_{50}	1.4	
Hg	haddock larvae, *Melanogrammus aeglefinus*	96h LC_{50}	0.098	Eisler 1987

Table 5-3

Effects of Inorganic Metals on Marine Fish.

Metal	Organism	Duration/Effect	Conc (mg/L)	Reference
	mummichog adults, *Fundulus heteroclitus*	96h LC_{50}	0.80	Eisler & Hennekey 1977
Ni	mullet adults, *Chelon labrosus*	96h LC_{50}	118.3	Taylor et al. 1985
Pb	mullet adults, *Chelon labrosus*	96h LC_{50}	>4.5	Eisler 1988b
	mummichog, *Fundulus heteroclitus*	96h LC_{50}	0.315	
	plaice, *Pleuronectes platessa*	96h LC_{50}	180	
Se	sheepshead minnow juveniles, *Cyprinodon variegatus*	96h LC_{50}	6.7	Heitmuller et al. 1981
Sn	sheepshead minnow juveniles, *Cyprinodon variegatus*	96h LC_{50}	58	Taylor et al. 1985
	dab adults, *Limanda limanda*	96h LC_{50}	>0.04	
Zn	mullet juveniles, *Aldrichetta forsteri*	96h LC_{50}	11.5	Negilski 1976
	hardyhead pre-adults, *Atherinasoma microstoma*	96h LC_{50}	40.5	
	mullet adults, *Chelon labrosus*	96h LC_{50}	21.5	Taylor et al. 1985
	mummichog adults, *Fundulus heteroclitus*	96h LC_{50}	60.0	Eisler & Hennekey 1977

Table 5-4

Effects of Inorganic Metals on Marine Invertebrates.

Metal	Organism	Duration Effect	Conc	Reference
Ag	bay scallop juveniles, *Argopecten irradians*	96h LC_{50}	35.8µg/L	Nelson et al. 1976
	mysid shrimp postlarvae, Mysidopsis bahia	96h LC_{50}	249µg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	19µg/L	
As	American oyster eggs, *Crassotrea virginica*	96h LC_{50}	7.5mg/L	Eisler 1988a
	amphipod, *Corophium volutator*	96h LC_{50}	16mg/L	Bryant et al. 1985
	bay scallop juveniles, *Argopectens irradians*	96h LC_{50}	3.49µg/L	Nelson et al. 1976
	blue mussel, *Mytilus edulis*	3-16d, lethal	16mg/L	Eisler 1988a
	clam, *Macoma balthica*	5°C, 90h LC_{50} 15°, 95h LC_{50}	 1g/L 125mg/L	Bryant et al. 1985
	copepod, *Acartia clausi*	96h LC_{50}	0.51mg/L	Eisler 1988a
	copepod juveniles, *Eurytemora affinis*	reduced survival	0.1mg/L	
	E. affinis adults	reduced survival	1mg/L	
	dungeness crab zoea, *Cancer magister*	96h LC_{50}	0.23mg/L	
	mysid shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	1.74mg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	893µg/L	
	oligochaete *Tubifex costatus*	96h LC_{50}	800mg/L	Bryant et al. 1985
	Pacific oyster embryos, *Crassotrea gigas*	96h LC_{50}	0.33 mg/L	Eisler 1988a
Cd	bay scallop juveniles, *Argopecten irradians*	96h LC_{50}	1.48mg/L	Nelson et al. 1976
		42d EC_{50} growth	78µg/L	Pesch & Stewart 1980
	grass shrimp, *Palaemonetes vulgaris*	96h LC_{50}	0.76mg/L	

Table 5-4

Effects of Inorganic Metals on Marine Invertebrates.

Metal	Organism	Duration Effect	Conc	Reference
Cd	hermit crab adults, *Pagurus longicarpus*	96h LC_{50}	0.32mg/L	Pesch & Stewart 1980
	mudsnail adults, *Nassarius obsoletus*	96h LC_{50}	35.0mg/L	Eisler & Hennekey 1977
	mysid shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	110µg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	15.2µg/L	
	sandworm adults, *Nereis virens*	96h LC_{50}	9.3mg/L	Eisler & Hennekey 1977
	softshell clam adults, *Mya arenaria*	96h LC_{50}	2.5mg/L	
	starfish adults, *Asterias forbesi*	96h LC_{50}	7.1mg/L	
Cr	mysid shrimp postlarvae, Mysidopsis bahia	96h LC_{50}	2.03mg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	132µg/L	
Cu	crab larvae, *Carcinus maenas*	48h LC_{50}	0.6mg/L	Connor 1972
	C. maenas adults	48h LC_{50}	109.0mg/L	
	lobster larvae, *Homarus gammarus*	48h LC_{50}	0.1-0.33mg/L	
	mysid shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	181µg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	55µg/L	
	shrimp larvae, *Crangon crangon*	48h LC_{50}	0.33mg/L	Connor 1972
	C. crangon adults	48h LC_{50}	29.5mg/L	
Hg	American oyster larvae, *Crassostrea virginica*	48h LC_{50}	5.6µg/L	Eisler 1987
	C. virginica adults	48h LC_{50}	5.5-10.2µg/L	

Table 5-4

Effects of Inorganic Metals on Marine Invertebrates.

Metal	Organism	Duration Effect	Conc	Reference
	bay scallop juveniles, *Argopecten irradians*	96h LC_{50}	89.0µg/L	Nelson et al. 1976
	blue mussel, *Mytilus edulis*	96h LC_{50}	5.8µg/L	Eisler 1987
	copepod adults, *Acartia tonsa*	96h LC_{50}	10.0-15.0µg/L	
	crab larvae, *Carcinus meanas*	48h LC_{50}	14µg/L	Connor 1972
	C. maenas adults	48h LC_{50}	1.2mg/L	
	dungeness crab zoea, Cancer magister	96h LC_{50}	6.6µg/L	Glickstein 1978
	fiddler crab zoea, *Uca pugilator*	8d LC_{50}	1.8µg/L	Eisler 1987
	hardshell clam larvae, *Mercenaria mercenaria*	48h LC_{50}	4.8µg/L	
	hermit crab adults, *Pagurus longicarpus*	96h LC_{50}	50µg/L	Eisler & Hennekey 1977
	lobster larvae, *Homarus gammarus*	48h LC_{50}	33-100µg/L	Connor 1972
	mudsnail adults, *Nassarius obsoletus*	96h LC_{50}	32.0mg/L	Eisler & Hennekey 1977
	mysid shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	3.5µg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	1.2µg/L	
	M. bahia	decreased reproduction	1.6µg/L	Gentile et al. 1983
	oyster larvae, *Ostrea edulis*	48h LC_{50}	1-3µg/L	Connor 1972
	O. edulis adults	48h LC_{50}	4.2mg/L	
	Pacific oyster embryos, *Crassostrea gigas*	48h EC_{50} abnormal development	5.7µg/L	Glickstein 1978
	polychaete larvae, *Capitella capitata*	96h LC_{50}	14.0µg/L	Eisler 1987
	prawn postlarvae, *Penaeus indicus*	96h LC_{50}	15.3µg/L	
	portozoan, ciliate *Uronema marinum*	24h LC_{50}	6.0µg/L	

Table 5-4

Effects of Inorganic Metals on Marine Invertebrates.

Metal	Organism	Duration Effect	Conc	Reference
	sandworm adults, *Nereis virens*	96h LC_{50}	70µg/L	Eisler & Hennekey 1977
	shrimp larvae, *Crangon crangon*	48h LC_{50}	0.01mg/L	Connor 1972
	C. crangon adults	48h LC_{50}	5.7mg/L	
	slipper limpet larvae, *Crepidula fornicata*	96h LC_{50}	60.0µg/L	Thain 1984
	C. fornicata adults	96h LC_{50}	330.0µg/L	
	softshell clam, *Mya arenaria*	168h LC_{50}	4.0µg/L	Eisler & Hennekey 1977
	starfish adults, *Asterias forbesi*	96h LC_{50}	60µg/L	
Mn	mussel larvae, *Mytilus edulis*	48h EC_{50}	30mg/L	Morgan et al. 1986
Mo	mussel larvae, *Mytilus edulis*	48h EC_{50}	147mg/L	
Ni	copecod, *Nitrocra spinipes*	96h LC_{50}	6mg/L	Carey 1981
	hermit crab adults, *Pagurus longicarpus*	96h LC_{50}	47mg/L	Eisler & Hennekey 1977
	mudsnail adults, *Nassarius obsoletus*	96h LC_{50}	72mg/L	
	mysid shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	508µg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	93µg/L	
	polychaete adults, *Nereis virens*	96h LC_{50}	25mg/L	Eisler & Hennekey
	softshell clam, *Mya arenaria*	168h LC_{50}	320mg/L	
	starfish adults, *Asterias forbesi*	96h LC_{50}	150 mg/L	
Pb	amphipod, *Ampelisca abdita*	96h LC_{50}	547µg/L	Eisler 1988b
	blue mussel adults, *Mytilus edulis*	96h LC_{50}	>0.5g/L	
	M. edulis larvae	96h LC_{50}	476µg/L	
	dungeness crab, *Cancer magister*	96h LC_{50}	575µg/L	

Table 5-4

Effects of Inorganic Metals on Marine Invertebrates.

Metal	Organism	Duration Effect	Conc	Reference
	mysid shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	3.13mg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	25µg/L	
	protozoan, *Cristigera* sp.	12h reduced growth	150µg/L	Eisler 1988b
	sand worm, *Neanthes arenaceodentata*	decreased reproduction	3.1mg/L	
	sea urchin embryos, *Anthocidaris crassispina*	48h, decreased development	2.2 mg/L	Eisler 1988b
	shrimp, *Crangon crangon*	96h LC_{50}	375mg/L	
Se	dungeness crab zoea, *Cancer magister*	96h LC_{50}	1.04mg/L	Glickstein 1978
	Pacific oyster embryos *Crassostrea gigas*	48h EC_{50} abnormal development	>10mg/L	
Zn	crab larvae, *Carcinus maenas*	48h LC_{50}	1.0mg/L	Connor 1972
	C. maenas adults	48h LC_{50}	14.5mg/L	
	hermit crab adults, *Pagurus longicarpus*	96h LC_{50}	0.4mg/L	Eisler & Hennekey 1977
	mudsnail adults, *Nassarius obsoletus*	96h LC_{50}	50.0mg/L	
	mysis shrimp postlarvae, *Mysidopsis bahia*	96h LC_{50}	499µg/L	Lussier et al. 1985
	M. bahia	decreased reproduction	166µg/L	
	sandworm adults, *Nereis virens*	96h LC_{50}	8.1mg/L	Eisler & Hennekey 1977
	softshell clam, *Mya arenaria*	168h LC_{50}	7.7mg/L	
	starfish adults, *Asterias forbesi*	96h LC_{50}	39.0mg/L	

Table 5-5

Effects of inorganic metals on marine benthic algae and phytoplankton.

Metal	Organism	Duration/Effect	Conc	Reference
As^{3+}	algae, 3 spp.	reduced growth	19-22µg/L	Eisler 1988a
	Champia parvula (Rhodophyta)	no sexual reproduction	95µg/L	Thursby & Steele 1984
		100% mortality	300µg/L	
	Plumaria elegans	18h, arrested development	0.58mg/L	
As^{5+}	algae, 2 spp.	no mortality	1g/L	
	Champia parvula	no sexual reproduction	10mg/L	Thursby & Steele 1985
	phytoplankton	4d, reduced biomass	75µg/L	Eisler 1988a
	Skeletonema costatum	reduced growth	0.13mg/L	
	Thalassiosira aestivalis	reduced chlorophyll *a*	75µg/L	
Cd	*Phaeodactylum tricornutum*	decreased growth	10-25µg/L	Eisler 1985
	Skeletonema costatum	decreased growth	10-25µg/L	
	Mediterranean species	48h, reduced production	1mg/L	Ibragim & Patin 1976
Cu	Mediterranean species	24h, reduced production	100µg/L	
		24h, increased production	1µg/L	
Hg	Mediterranean species	48h, reduced production	10µg/L	
		48h, increased production	1µg/L	
Pb	*Phaeodactylum tricornutum*	96h LC_{50}	>5mg/L	Eisler 1988b
	Skeletonema costatum	12d, 50% growth inhibition	5.1µg/L	
	phytoplankton, (mixed)	4d, reduced biomass	21µg/L	

comparisons can be made. In addition, acute values are often indicative of chronic values. Kenaga (1982) found 86% of LC_{50} acute values to be less than two orders of magnitude different from chronic NOEC concentrations for the same chemical and fish or invertebrate species.

Table 5-3 summarizes the effects of metals on marine fish. From these data, Ba and Ni appear to have relatively low toxicity, whereas Cu and Hg are highly toxic. Most of the 96 h LC_{50} values for the other metals range between 10 and 60 mg/L, with the notable exception of Pb which had widely ranging toxicity to different species. Though this review does not provide evidence regarding sensitivity of early life stage fish relative to juveniles and adults, other reviews and freshwater research indicate a strong tendency for lower tolerance of younger fish to inorganic and organic chemicals, particularly after gill development (Macek and Sleight 1977, McKim 1977, Ward and Parrish 1980).

Concentrations of metals that were acutely toxic to invertebrates (Table 5-4) had a comparatively greater range than for fish, yet the relative lower tolerance of invertebrates to metals is clear. The most consistently toxic metals to invertebrates were Ag, Hg, and Pb. Typically, adult mortality occurred at 10 to 100x the concentration lethal to larvae. Chronic effects, such as decreased reproduction of mysid shrimp, were invariably more sensitive than 96 h LC_{50} values of post-larvae (Lussier et al. 1985). Out of nine metals investigated, mysid shrimp acute values ranged from 1.9x to 125x greater than chronic values for As and Pb, respectively. Generally, acute toxicity occurred at 3-10x the concentrations causing chronic toxicity with this species. In support of this, Sanders (1986) concluded that estuarine zooplankton were resistant to acute effects of As, but that reproductive success was affected at much lower levels.

Regarding phytoplankton, the increased production observed by Ibragin and Patin (1976) (Table 5-5) resulting from exposure to low levels of Hg and Cu was at metal concentrations similar to Mediterranean surface seawater. The influence of metal valence state on toxicity is shown with As, which marine phytoplankton have been shown to reduce and methylate (Sanders 1983). After long term As exposure, Sanders and Cibik (1985) noted changes in phytoplankton species composition compared to controls. Stauber and Florence (1985) showed Mn to alleviate Cu toxicity to the marine diatom (*Nitzschia closterium*).

Comparisons between Tables 5-3, 5-4, and 5-5 reveal some general trends. Similar sensitivity of phytoplankton and invertebrates to As is apparent, whereas adult fish were more tolerant. Similarly, fish were more tolerant to Cd, Cr, and Cu than invertebrates. Toxicity of metals to microbiota was not summarized since microbiota are usually very insensitive to toxicants due to a variety of evolved resistance mechanisms (Wood and Wang 1983). Rarely are microbiota adversely affected before higher organisms (Pritchard and Bourquin 1985). Microbiota are more likely to be affected by smothering and abrasion resulting from STD rather than toxicity.

A review by Rand (1985) of avoidance responses of fish and invertebrates to metals and other chemicals indicated, as is typical, generalizations cannot be drawn. Much of the review was in reference to freshwater species. Regardless, it was found that some organisms avoid sublethal concentrations of some chemicals or complex effluents while others species do not avoid lethal concentrations.

Interactive toxicity of metals is likely to be a major factor in most real world studies. Laboratory experiments illustrate some of the multitude of mixtures and responses. Brown and Dalton

(1970) exposed rainbow trout (*Oncorhynchus mykiss*) to Cu, Zn, and Ni and found the toxic contributions to be additive. Finlayson and Verrue (1982) found combinations of Cu, Zn, and Cd to exhibit additive or antagonistic toxicity to chinook salmon (*Oncorhynchus tshawytscha*) depending on the ratios of the metals in solution. In contrast, the effects of Cu and Zn on survival of mummichog (*Fundulus heteroclitus*) were synergistic (Eisler and Gardner 1973). Benijts-Claus and Benijts (1975) found Pb and Zn, individually, to delay development in the estuarine mudcrab (*Rhithropanopeus harrisii*), but to have no adverse effect when tested in combination, therefore, in this instance, Pb and Zn were antagonistic.

As with toxicity, bioaccumulation is greatly affected by chemical interactions. Eisler and Gardner (1973) found residues from surviving mummichog (*Fundulus heteroclitus*) exposed to Cd and Zn not to conform to patterns observed for fish exposed to the single elements. Similarly, Phillips (1976b) found Cu uptake by *Mytilus edulis* to be influenced by the presence of other metals, whereas Zn, Cd, and Pb were not. Iron was found to inhibit Pb uptake by the deposit feeding bivalve *Scrobicularia plana* (Luoma and Bryan 1978). The variable nature of metal accumulation is also evident with regard to depuration. For example, Okazaki and Panietz (1981) showed that different metals depurate at different rates from oysters (*Crassostrea gigas*, *C. virginica*).

Various other water characteristics, as well as the presence of organic contaminants, influence metals accumulation and toxicity. Organic contaminants may impact metal detoxification by binding to metallothioneins in the organism (Jenkins 1984). Metallothioneins are proteins which are largely responsible for metal detoxification *in vivo*. Naturally occurring organic compounds in water may decrease metal availability due to complexation. The effects of natural chelators have been demonstrated repeatedly, for example, Crecelius et al. (1982) found Cu availability to marine bivalves to decrease due to complexation by natural and artificial organic chelators. As mentioned, other natural variables also influence bioaccumulation. Bryan and Hummerstone (1973) found that Mn concentrations in a polychaete (*Nereis diversicolor*) decreased with increasing salinity. Skaar et al. (1974) found phosphate to influence Ni uptake by a diatom. Lewis et al. (1972) found Cu toxicity to a calanoid copepod (*Euchaeta japonica*) to be reduced by the presences of diatoms, clay particles, ascorbic acid, sewage effluent, humic acid, and soil extracts.

The uptake potential of metals is also highly variable between species of organisms (Ray et al. 1981). Peddicord (1980) monitored uptake of metals by coast mussels (*Mytilus californianus*), spot-tailed sand shrimp (*Crangon nigromaculata*), and dungeness crab (*Cancer magister*) exposed to suspended dredged sediment contaminated with organic and inorganic chemicals. Coast mussels accumulated Cu and Pb, and to a slight degree, Zn, but did not accumulate As, Cd, Fe, Mn, Hg, or Ni. Of the preceding list of metals, spot-tailed sand shrimp accumulated As, Cd, Cu, Mn, and Zn, and dungeness crab accumulated Mn, Cd, Pb, Ni, and Zn. Penrose et al. (1975) found sea urchin (*Strongylocentrotus droebachiensis*) to be the only organism of eight taxa, including mussel (*Mytilus edulis*) to accumulate As from mine drainage. Amiard et al. (1987) demonstrated that decapod crustaceans and fish were better regulators of Cu and Zn than bivalves and polychaetes. Sims and Presley (1976) found mollusks (*Crassostrea virginica*, *Rangia cuneata*) to have higher tissue metal levels than crustaceans or fish sampled from the same bay. Rosemarin et al. (1985) showed *Fucus* sp. to accumulate more As than mussels, polychaetes, gastropods, and crustaceans.

In general, the common mussel (*Mytilus edulis*) is considered a good indicator organism for metals concentration and availability (Phillips 1976a, 1976b, Turgeon and Lauenstein 1991), though Preston et al. (1972) found *Fucus* sp. to be a better indicator, noting higher variability in *M. edulis* metal levels. Similarly, Ward et al. (1986) found seagrass (*Posidonia* spp.) to be good indicators of metal contamination, as are marine algae (Wong et al. 1979, Jennet et al. 1983, Yamaoka and Takimura 1986).

Reports of biomagnification in marine food webs are rare. While the phenomenon can be demonstrated, as done for As by Wrench et al. (1979) in a three step laboratory marine food chain, other investigators have found no evidence of biomagnification in field studies. Ward et al. (1986) found no biomagnification in seagrasses and 23 species of marine animals near a Pb smelter, and concluded that primary producers had higher metal levels than consumers. In a review, Young (1984) cited no examples of biomagnification of metals in marine food webs, except Hg (see discussion). In reference to the scarcity of evidence for biomagnification, the author cited the complicated and variable nature of marine food webs. Specific to coastal mining discharge, Ellis (1984) reported that no evidence to date of metal biomagnification exists. Harding (1983) drew the same conclusion in reference to STD.

5.5.4 Milling Reagents Research

Few reports were found specifically addressing the toxicity or accumulation of metal mine milling reagents. Other than for cyanide (CN), most data were from a single freshwater study of the toxicity of collector reagents to rainbow trout *(Oncorhynchus mykiss)* (Fuerstenau et al. 1974). Due to the paucity of information, these data are included in this review for the purpose of relative toxicity comparisons, though reliable extrapolations to saltwater cannot be made. The data are summarized in Table 5-6. The experiments were performed at pH 8.6 and water hardness as $CaCO_3$ = 203 mg/L. A review by Davis et al. (1976) and studies by Leduc et al. (1974 and 1975) addressed freshwater toxicity of milling reagents for various species, furnishing some evidence of a general lower tolerance of invertebrates to milling reagents than fish. These data were not included in this report since the variety of species tested did not allow for relative toxicity comparisons of reagents.

In viewing Table 5-6, it is apparent that xanthates and carbamates were the most toxic of the reagents studied by Fuerstenau et al. (1974), and toxicity decreased as temperature decreased for the two chemicals for which temperature comparisons were made. The concentrations of these reagents in tailings from operations studied by Read and Manser (1975, 1976) were considerably lower than lethal concentrations. Potassium amyl xanthate, oleic acid, and sodium carboxymethyl cellulose were considered biodegradable, whereas isopropyl ethyl thiocarbamate, sodium di-secondarybutyl dithiophosphate, and polyglycolether (Dowfroth 250) were not. Degradation products were not mentioned. The latter two results in Table 5-6 are from other studies using freshwater salmonids. It appears that the milling byproduct, NH_3, was probably more toxic than the other reagents, though this involves an inter-species comparison (Servizi and Gordon 1990). HCN was unquestionably the most toxic of the reagents listed to freshwater rainbow trout (Leduc 1984) and is generally the most toxic component of cyanide containing mill effluent (Leduc et al. 1975, Servizi and Gordon 1990, Palmes 1993).

Sodium cyanide additions in the milling process result in various compounds including cyanate (CNO^-), thiocyanate (SCN^-), and metallocyanides. Cyanide destruct processes (Weathington

Table 5-6

Acute toxicity of metal mine milling reagents to freshwater rainbow trout (*Oncorhynchus mykiss*) at 12°C [1].

Chemical	96h LC_{50} (mg/L)
Sodium ethyl xanthate (Aeroxanthate 325)	14-16
Sodium ethyl xanthate (Z-4)	13-15
Potassium ethyl xanthate (Z-3)	15-17
Pure potassium ethyl xanthate	1.8-2.0
	15-20 at 8°C
Sodium isopropyl xanthate (Aeroxanthate 343)	18-20
Sodium isopropyl xanthate (Z-11)	18-20
Potassium amyl xanthate (Aeroxanthate 350)	70-80
Pure potassium anyl xanthate	70-75
Sodium Aerofloat	400-410
	310-330 at 16°C
Pure potassium diethyl dithiophosphate	800-825
Aerofloat 238	600-625
Pure potassium disecondarybutyl dithiophosphate	>1000
Isopropyl ethylthionocarbamate	45-48
Pure isopropyl ethylthionocarbamate	40-45
Hydrogen cyanide (HCN)	42µg/L [2]
Ammonia (NH_3)	450µg/L [3]

1 Fuerstenau et al. 1974

2 Leduc 1984 (similar condition as Fuerstenau 1974).

3 Servizi and Gordon 1990 (juvenile chinook salmon (*Oncorhynchus tshawytscha*) at 7°C, water hardness = 84mg/L).

1988, Scott 1989, McGill and Comba 1990) typically reduce much of the cyanide to NH_3 as mentioned above. Thompson (1984) determined the 14 d LC_{20} for mussels (*Mytilus edulis)* to be 100 μg/L HCN, and found growth to be reduced after 14 d exposure to 18 μg/L HCN. The 96 h LC_{50} for mysid shrimp (*Mysidopsis bahia*) was 113 μg/L, and reproduction was reduced at >43 μg/L HCN (Lussier et al. 1985). In a review, Eisler (1991) stated that fish are the most sensitive aquatic organism to cyanide, particularly juveniles and adults. This is in contrast to most chemicals which are more toxic to early life stages and to invertebrates. Also cyanide is unusual in that it is more toxic at temperatures near freezing (Palmes 1993). Despite the apparent high potential for acute toxicity, Eisler (1991) concluded that cyanides are not mutagenic, teratogenic, or carcinogenic and do not persist in aquatic environments.

6.0 DISCUSSION

Reports from the Greenland STD operation and the two British Columbia examples provide some of the needed information regarding the consequences of increased metals, suspended sediment, and sediment deposition to biota. In the following discussion, an attempt will be made to synthesize these reports, along with data from related research, to support conclusions and fill in informational gaps relating to biological consequences of STD. Because of the overriding economic and regulatory concerns of STD, a relevant biological consequence, in this context, applies mainly to commercially important species. It is beyond the scope of this discussion to consider the value of less tangible considerations. Any impact on biological production, however, will be addressed since it could have implications to all resident biota.

6.1 Consequences of Submarine Tailings Disposal

6.1.1 Acute Effects

As indicated in Table 5-2, the minimum suspended sediment concentration causing mortality to fish or invertebrates after acute exposure was approximately 1.5 g/L (Sherk et al. 1975), approximately 100x greater than concentrations measured at STD sites. Mackas and Anderson (1986) reported suspended sediment concentrations associated with the AMAX/Kitsault STD operation of 10-20 mg/L near the discharge, comparable to peak near surface sediment loads introduced by rivers in the same inlet. Hesse and Reim (1993) reported a suspended sediment concentration of 12 mg/L, 18 m above the bottom at the Island Copper operation. In concurrence with these findings, lethal concentrations of suspended red mud (Al refining waste) (Blackman and Wilson 1973) and dredged sediment (Salo et al. 1979) were considerably higher than levels of suspended sediment likely to occur from STD.

Goyette and Nelson (1977) commented that salmon must pass through areas containing resuspended tailings from the Island Copper operation. This is unlikely to cause a direct effect. Salo et al. (1979) demonstrated tolerance by chum salmon (*Oncorhynchus keta*) to high levels of suspended dredged sediment (Table 5-2). Furthermore, rivers high in suspended glacial flour are typically traversed by all North American species of Pacific salmon (Meehan 1961).

The likelihood that increased metals at the Greenland site were bioavailable and could have caused acute effects is indicated by the accumulation in biota. For this reason, the bioaccumulation data may be the most relevant chemical information gained from the studies reported for this mine, since all of the water and sediment chemistry data were in reference to total metal concentrations with no indication of bioavailability. Some of the Greenland seawater samples contained metal levels higher than those shown to cause acute or chronic effects to fish and invertebrates for Pb or Zn alone (Connor 1972, Eisler and Hennekey 1977, Lussier et al. 1985, Eisler 1988b), in addition to possible interactions with other metals. It is entirely possible that there were decreases in survival, as well as chronic effects, as a direct result of leached metals from the Black Angel waste rock and tailings, though reported data did not address this.

In general, if toxicity from metal exposure were to occur, it would most likely take place in benthic organisms due to the higher concentration of metals in deposited sediment (Goyette et al. 1985) and the relative immobility of benthos relative to water column animals. This risk would be greatest for organisms with a benthic early life stage. With the exception of the Black Angel mine, acute effects due to metals resulting from STD are unlikely, though the bioassay results of the Quartz Hill tailings demonstrated that the potential exists (E.V.S. 1984, Mitchell et al. 1985). As mentioned, the latter experiments represented an unrealistic worst case scenario, though such experiments are worthwhile since negative results would eliminate the need for more complicated flow-through exposures.

Redeposited tailings may become exposed to oxidizing conditions in the intertidal region. This region is one of the least desirable locations for tailings because it maximizes the likelihood for leaching of metals. Depending on the composition of the tailings, such an occurrence could result in release of metals into water and sediment with possible biological consequences. Surveys of benthos in tailings initially dumped in the intertidal zone may serve as examples of what could occur if tailings from an STD operation are resuspended.

Portions of Juneau, Alaska are built on mine waste rock and tailings, and a recreational beach is comprised entirely of tailings from a gold mine which ceased operation in the early part of this century. If toxicity occurred, effects from such an aged deposit are likely past. The previously discussed Chile operation serves as a more modern example. Tailings at the Chile intertidal dump site were acutely toxic to benthos and fish (Castilla and Nealler 1978). Yet, it is quite likely that the solubility of metals from these tailings would have precluded the option of modern STD in North America. Still, the Chile example, coupled with the acknowledged possibility of tailings resuspension, does justify studies of tailings in a worst case scenario, should unpredicted intertidal deposition occur.

Regarding milling reagents, unidentified fish were detected in the Island Copper mine turbidity current (Hay et al. 1982). If fish are attracted to turbidity currents, an attempt to identify the species of fish and duration of exposure should be made. It is possible that they may not instinctively avoid plume associated contaminants which may be present, particularly in the near-field. There is a lack of information available regarding the marine toxicity of milling reagents and their breakdown products. Though mixing zones are often permitted so that effluents can meet discharge criteria, biota within mixing zones are transient and consequences of exposure should be determined. Available data indicate that the near-field constituents, including cyanide, are likely of little large scale consequence owing mainly to rapid dilution and/or degradation (Kessler/E.V.S. 1992).

In reference to complex interactive toxicity, Sloterdijk et al. (1989) concluded that the chemistry of elutriates was not predictive of the toxic potential of contaminated sediments. Elutriates in this instance were the resulting supernatant after vigorous mixing of sediment with water. Similarly, Peddicord (198J) stated that the toxicity of San Francisco Bay sediment could not be predicted, possibly because of synergism, but more likely due to toxicity of a component not chosen for analysis. The author did, however, isolate the toxic effects of suspended sediment as being chemical in nature and not due to physical effects of the particles. Ahlf et al. (1989) evaluated a number of bioassay techniques for assessing the causes of biological effects based on analysis of sediment contaminants and concluded that biological responses could not be predicted.

In order to fully understand the biological significance of contaminated sediment, chemical characterization, laboratory bioassays, and on-site infaunal sampling should be performed (Long and Chapman 1985, Chapman 1990). In an effort to isolate sources of toxicity, methods have been developed to identify toxic components of effluents and sediments (Samolloff et al. 1983, Mount and Anderson-Carnahan 1988, Ankley et al. 1992). Clearly, the most definitive means for evaluating potential toxicity of STD discharges are whole effluent bioassays, *in situ* when possible. If deemed toxic, some of the previous techniques may allow isolation of the sources of toxicity.

Surveys at the Island Copper operation verified a reduction of benthic fauna due to smothering (Ellis and Heim 1985, Ellis and Hoover 1990b). Because of tailings resuspension and deposition, smothering was also observed in some shallow water locations. Benthic larvae carried on currents from unimpacted areas (Chandler and Fleeger 1983) may also be smothered if they are forced to settle in STD depositional areas, therefore, local dispersion patterns of meroplankton are worthy of investigation. Smothering will likely be more widespread if a site has low natural sedimentation since the resident biota will be less tolerant than an assemblage in an area of high natural sedimentation (McFarland and Peddicord 1980).

6.1.2 Chronic and Secondary Effects

Ellis and Heim (1985) noted an absence of zooplankton in waters with visibly high levels of suspended sediment at the Island Copper operation. Sediment may be more likely to stay in suspension if zooplankton abundance is reduced since zooplankton contribute significantly to sediment settling through ingestion and pelletization (Syvitski 1980, Hu 1981). Overall, though some effects of suspended tailings on plankton have been reported (Parsons et al. 1986a, 1986b), impacts appear to be minor (Anderson and Mackas 1986, Ellis 1989).

STD will reduce benthic standing crop while the operation is active (Brenchley 1981, Turk and Risk 1981, Ellis and Heim 1985). The possible effects that this may have on pelagic production are difficult to predict. Graf (1992) stated that benthic remineralization of organic particles is the basis for new production in the euphotic zone. Smetacek et al. (1984) described an annual cycle of an inshore pelagic system at a British Columbia latitude. During summer stratification, pelagic production was based largely on regenerated nutrients. This implied that increased inorganic sedimentation resulting from STD may reduce not only benthic production, but pelagic production as well. During spring blooms typical of British Columbia and Alaska latitudes, the majority of phytoplankton settle to the bottom unconsumed. If inorganic sedimentation is high due to STD, this organic material may be buried prior to remineralization. Microbia will likely not be present in significant quantities to remineralize detritus due to smothering and abrasion caused by sediment transport (Jumars and Nowell 1984) and as a result, natural periods of nutrient limitation in the water column may be amplified.

6.1.3 Bioaccumulation

If bioaccumulation were to occur, it would most likely take place in benthic organisms for the same reasons noted for toxicity. The potential for bioaccumulation could be affected by the natural sediment layer which will eventually cover the tailings, the species which recolonize and burrow in the new substrate, and the multitude of other variables present on the sea floor.

Incidents of human poisoning, such as occurred in the 1950s in Japan due to consumption of Hg contaminated fish and shellfish from Minimata Bay (Forstner and Wittmann 1979, Ellis 1988), justly raised concerns over bioaccumulation of toxins. Much of the ensuing research, however, simply reported concentrations of various chemicals in various organisms, with no indication of the ecological relevance. Determination of accumulation is essentially meaningless unless correlations to ecosystem effects can be made, or if the organism is consumed by humans. In order to correlate bioaccumulation with ecosystem level effects, relations between metal content and biota fitness or abundance must be made. Peddicord (1984) provides an excellent discussion on this topic.

The implications of bioaccumulation at the Black Angel deposit were not apparent, with the exception of determination of metal levels in some commercially important species. For example, halibut and cod accumulated insignificant metal burdens from the discharge (Johansen et al. 1991). The lack of metal uptake by these fish species is of considerable worth in regard to human health concerns and can likely be generalized to other mobile fish species. The elevated metal levels in the water and sediment, however, could have had serious implications in a more populated area depending on the commercial species present. In comparison to the Black Angel operation, bioaccumulation from the Island Copper and AMAX/Kitsault mines has been minimal and probably of no significant biological consequence, illustrating the importance of preliminary leaching studies. The literature pertaining to bioaccumulation, as with toxicity, illustrates the major role of chemical interactions and the need for site specific research.

6.1.4 Behavioral Effects

The most likely direct effects of suspended tailings on fauna are avoidance and changes in feeding behavior of some fish species. Avoidance and/or decreased feeding by herring (*Clupea harengus harengus*) and smelt (*Osmerus mordax*) (Table 5-2) occurred at levels of suspended sediment within ranges resulting from STD.

The only conceivable effect on salmon of suspended sediment resulting from STD is disorientation with regard to locating their natal streams. Salmon home stream detection is at least partially a combination of visual and olfactory cues (Healey 1991), yet it is not completely understood, and a perturbation may interfere with critical cues needed for stream location. No evidence, however, was found indicating that this has occurred. Salmon escapement data for a stream entering Rupert Inlet did not indicate impact in this instance (Goyette and Nelson 1977). Overall, properly engineered STD discharges appear to have little potential for damage to salmon stocks.

6.1.5 Habitat Alteration

Habitat alteration due to redeposition of Island Copper tailings in shallow and formerly rocky areas may have more significant biological consequences than the tailings in suspension or at the fjord bottom. Numerous species rely on coarse grained or rocky substrates in shallow and deep water. A reduction in these habitats is likely to cause a concomitant reduction in the abundance of these animals. Many commercially important species rely on coarse grained substrate, including rockfish, and some flatfish and crab species (DeGroot 1980, Rosenthal et al. 1982, Krieger 1993). Goyette and Nelson (1977) concluded that resuspension and deposition of Island Copper tailings has resulted in a permanent alteration of habitat, precluding recolonization by organisms which require a relatively sediment-free substrate.

The importance of understanding tailings behavior was brought forth by the Island Copper resuspensions. The depositional outcome was determined by chance, and in this case or in future instances, may lead to obliteration of breeding grounds, foraging areas, and refugia for some species. Short-term surface water impacts of suspended sediment at the proposed Quartz Hill STD project were said to be beyond predictability (Hesse and Reim 1993). If areas receiving deposition were similarly unpredictable, some species may have been driven out due to habitat alteration. The altered bathymetry and often significant reductions in the depth of STD fjords may also have ecological implications, though these possibilities remain unexplored.

6.2 Ecological Recovery After STD

The rate of benthic recovery is likely to limit ecosystem recovery as a whole after an STD operation. Tailings may be unsuitable for some species due to metal content, grain size, or low organic content (Fricke and Flemming 1983, DeWitt et al. 1988, Pawlik 1992). In such a case, recovery may be limited until a sufficient natural sediment layer has accumulated on the tailings. Given tailings which are equally well suited for colonization as natural sediment, the area will still have to be re-seeded with settling meroplankton and proceed through a series of successional stages (Taylor 1986). In comparison, pelagic organisms are far removed from any direct effects from the settled tailings, though they rely to an extent on the benthos for forage and nutrient recycling. Also, pelagic animals can repopulate more rapidly since they are often highly mobile and transient, or in the case of plankton, often have short generation times and are transported widely by currents.

Reports on recovery of smothered benthic areas vary (Kathman et al. 1983, Kathman et al. 1984, Ellis and Hoover 1990b), partly due to the difficulty in defining "recovery". It is difficult to measure recovery due to the variable and cyclical nature of assemblages (Johnson 1970, Arntz and Rumohr 1982, Winiecki and Burrell 1985). Lewis (1970) observed that populations of marine species fluctuate over time spans ranging from a few to 20 years. Even if recolonization is rapid, however, this does not necessarily indicate that benthic smothering should not be of concern. The duration of an STD operation can last decades and implications of the lost benthic production time, with possible pelagic implications, are worthy of consideration as discussed by Waldichuck (1978).

If the tailings are suitable, recolonization may be able to proceed even if organisms are accumulating metals from the tailings, as found by Parrish et al. (1989). In such a case, accumulation by human food organisms would still be a consideration. In contrast to the findings of Parrish et al. (1989), extensive sampling of polluted fjords by Rygg (1985) revealed a strong cause-effect relationship between sediment Cu and reduced benthos diversity, and weak correlations for Pb and Zn.

Benthic assemblages are likely a result of passive planktonic transport followed by passive and active modes of substrate selection (Butman 1987, Palmer 1988). It has been demonstrated that planktonic stages of benthos display preference for substrate on which to settle based on grain size, chemical cues, and microbial film, among other factors (Pawlik 1992). Cues that settling infauna respond to may be positive (e.g. presence of conspecifics) or negative (e.g. lack of food) (Woodin 1991). Tailings beds are relatively large and homogeneous due to the single source of the sediment and burial of natural substrate anomalies. As a result, settling meroplankton may not have the option to select a site of colonization, but rather will have to prosper or perish on the tailings until natural sediment covers them.

6.3 Research Needs

While an array of site-specific bioassays must be performed for each STD operation, the remaining general research needs include the following:

1. Development of a data base on the marine toxicity and bioaccumulation potential of milling reagents and their break-down products.

2. Development of *in situ* methodologies for pre-operational prediction of the bioaccumulation and recolonization potential of tailings deposits.

3. Further quantification and qualification of recolonizing benthic and pelagic assemblages at current and former STD sites.

4. Determination of the effects of tailings deposition on benthic and pelagic standing crop and production.

5. Determination of the relationship between tissue metal concentrations and organism fitness.

The first point addresses a fairly straightforward issue. While the potential effects of milling reagents present in a final effluent are probably minor compared to other considerations, it is not sensible to dispose of waste at the volumes and for the duration typical of an STD operation without a complete understanding of its constituents and possible consequences. Unimagined ecological implications have resulted from past introduction of chemicals into the environment. While the possibility of such an occurrence is remote, there is no need for assumptions when concerns can be quelled with simple bioassays.

Secondly, due to the vast number of chemical and physical variables in a natural system, *in situ* investigations, in addition to laboratory studies, should be performed whenever possible. *In situ* bioassays serve the obvious purpose of monitoring during an STD operation, but they also could be used as a predictive tool prior to permitting, eliminating some of the guessing and extrapolating. The recolonization and bioaccumulation potential of tailings deposits could be explored, possibly through use of the tray method described in Section 5.4 (Burrell 1981).

Recolonization should be investigated so that it may be factored into the total lost biological production time. Bioaccumulation with regard to human food organisms is worthy of investigation since bioaccumulation has been documented at modern STD operations, and because there is little room for error. A realistic determination of recolonization and bioaccumulation potential can only be made with the actual tailings, natural sedimentation, the complete population of resident biota, and all of the other natural variables, in other words, *in situ*.

Much could be learned from the Black Angel tailings deposit. It serves as a worst case scenario and offers substantial research opportunity. Particularly, ecosystem implications of bioaccumulation could be investigated, in addition to gaining information regarding benthic recolonization of contaminated sediment and understanding the relation of natural sediment deposition on top of reactive tailings to marine chemistry and biology.

The fourth point addresses the need for a better understanding of the interdependence of benthic and pelagic biota and the effect of benthic habitat alteration. The effect of benthic habitat alteration is straightforward and can be assessed on a site-specific basis with respect to spawning, foraging, and refuge habitat. The effects of altered bathymetry are not as simple to study, but should also be addressed. Knowledge of benthic-pelagic coupling is based on theory or closely linked portions of marine ecosystems. To actually apply this knowledge to larger organisms and species trophically distant from each other in a marine ecosystem is difficult, yet this is the only way to quantify the possible ramifications of STD smothering.

The final research need is common to any study of pollution and bioaccumulation. If biota are not to be consumed by humans, why be concerned with concentrations of substances in tissues of organisms? Relations between tissue concentrations and the ability of organisms to grow, reproduce, and survive must be established to understand the implications of bioaccumulation.

6.4 Conclusions

Assuming an ideal STD operation with negligible tailings reactivity, no resuspension of tailings, and disposal into a deep, soft-bottomed basin that is not critical habitat for commercially important species, the only likely major consequence to biota is smothering of benthos or relocation of epibenthos for the duration of the operation and recolonization period. The importance of benthic smothering depends on the species present, particularly harvested crabs, mollusks, and flatfish. If the above assumptions are not met, reductions in organism fitness may result due to increased metals in the water and sediment, also possibly leading to restrictions on human consumption of these organisms. Even if organisms accumulate metals, biomagnification is unlikely. If tailings are resuspended, coarse or rocky habitats may be buried with a consequent decrease in animals relying on these habitats for spawning, foraging, or refuge.

Inadequacies in biological STD research include the need for more information on milling reagents and their fates, pre-operational methodologies for predicting the consequences of STD, further study of existing submarine tailings deposits, a better understanding of the implications of benthic smothering and habitat alteration, and establishment of relations between bioaccumulation and biota fitness. Previous and current utilization of STD has resulted in some metal pollution and habitat alteration. These instances highlight the need for a more thorough understanding of non-biological aspects of STD, namely, tailings transport and chemistry.

Peer review is important for quality control in scientific reporting, especially in applied research where there may be financial cause for bias. Conclusions from this review were, in part, based on gray literature which contains much of the information needed to evaluate STD. When available, published research supported the conclusions, but as a whole, the argument that STD is of little biological consequence is weakened by the lack of published data pertaining to it.

Any perturbation of a water body on the scale of an STD operation will undoubtedly have biological implications. If a mining operation is going to proceed, the potential consequences of STD should be weighed against environmental degradation which would result from other tailings disposal options.

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